# START-UP

Start-up work has been going on since the first wind turbines were erected, and since the middle of November the workforce for this task is app. 20 men

The last wind turbine is erected in the middle of December 2000, and usually a month is needed to finish the job. However, a delay caused by damaged sea-cables means that we cannot start reducing the workforce until February.

The first phase of the start-up work consists of tightening all bolts to the correct torque. This is necessary where the tower sections are bolted together, and where the nacelle is fastened on top of the tower. The bolts fastening the blades to the hub, and the hub's fastening to the main shaft are also checked.

The electrical installations are completed. Thick aluminium tower cables are fastened and led to a junction box in the upper tower section, from where flexible cables are connected to the generator. A variety of control cables are led to the control system, and finally all electrical connections are tested.

Work proceeds according to a checklist: The generator is checked for alignment with the shaft. The hydraulic systems are checked for air. The oil levels in the main gearbox and yaw-gearbox are checked, and the nacelle yaw gear wheel is greased. The control systems are checked, and braking tests are carried out. The tower and the nacelle de-humidifiers are tested, and finally the wind turbine is cleaned.

## START

The first wind turbines are grid-connected on the 27th of December 2000, and on the 6th of March 2001 all 20 turbines are supplying electricity to the public grid.

After the first two-three days of operation the wind turbines are checked for signs of possible oil leakages. To ensure a shooth running-in of the gearbox, full load capacity during the first 500 hours of operation is limited to a maximum load of 1.4 MW.

# OFFSHORE MODIFICATIONS

The Bonus 2 MW wind turbine is basically developed for offshore wind farm use. The main difference between land based and offshore turbines is that the latter are more exposed to demanding climatic conditions and structural corrosion.

In addition the turbines can be difficult to access due to bad weather conditions, and finally an offshore site will involve greater expense with possible replacement of main components than tasks carried out on land.

The aim of a wind turbine modification for offshore sites is first and foremost an increased corrosion protection, reduction of maintenance requirements and an improved monitoring system.

## CORROSION PROTECTION

The corrosion protection of a Bonus 2 MW wind turbine can be divided into an exterior and an interior protection.

The exterior corrosion protection of the steel components on offshore sites consists of a coating system fulfilling the standards required for North Sea offshore erections. On shore, a normal standard coating system is applied in accordance with the required operational life according to the expected climatic conditions.

The surface of the fibreglass blades is similar to that of fibreglass boat hulls and therefore it requires no additional corrosion protection for offshore use.

Interior corrosion protection is adapted to offshore conditions, partly using improved coating systems and partly by maintaining a dry environment inside the wind turbine structure.

A precondition for a dry interior climate is that the interior is tightly sealed. There is no open cooling in the wind turbine. The gearbox and the generator are cooled by heat exchangers, recycling the air used in the cooling system. This arrangement is also used as a standard in the 2 MW models for land sites.

In order to maintain a low interior air humidity, offshore turbines are equipped with special dehumidifying devices placed in the tower and the nacelle. A correctly sized dehumidifier can maintain the inside relative humidity level below 60%, which is the limit for steel corrosion.

For additional protection, the main electric components (generator, control systems, etc.) are equipped with stand-by heating systems, preventing condensation even during sudden variations in temperature.

## SERVICE AND MAINTENANCE

Bonus has attempted to reduce the necessary service and maintenance parameters for the 2 MW wind turbine compared with normal requirements for large turbines.

The lubrication system has long service intervals. Many bearings, including the blade bearings, are automatically greased. A special gearbox oil filter, separated from the normal oil cooling system, ensures high oil purity. Gear oil temperature is maintained inside a narrow temperature range by a preheating and cooling system. The såme lubrication quality is maintained under all operational conditions.

Where possible, all bolted connections are carried out with extended pretensioning. This means that losses in pretension and the needs for re-tightening are reduced. Using appropriate statistical methods, bolt control checks can eventually be reduced to sample test control.

In the wind turbine safety systems, sensors are usually installed in parallel. In the event of failure of any sensor, the wind turbine is brought to a stop. On Bonus 2 MW turbines this concept is modified, so that failure of any single or several parallel sensors will not automatically result in an operational shutdown. It will only result in an error indication. However, in the actual safety system, normal operational procedures are not deviated from.

To ease the work during maintenance operations, the 2 MW wind turbine is equipped with two hydraulic cranes, mounted in the nacelle. These two built-in cranes lift tools and spare parts and can place them anywhere in the nacelle.

If these built-in cranes should lack lifting capacity, they can be used to install a large mobile crane, capable of lifting even the heavy main components (blades, gearbox, generator). Normally, one mobile crane is available for all the turbines on a large offshore wind farm.

The advantages of these reduced maintenance requirements also have an effect on land based wind turbines, and with the exception of the mobile crane, they are now standard in all Bonus 2 MW models.

## MONITORING

Bonus uses a computer network to monitor and operate offshore wind farms. The network is structured like networks in most companies. The installation of a computer in every wind turbine, each with it's own net address, allows communication to and from the wind turbine using a normal internet browser.

The network monitoring system has certain advantages compared to more traditional supervisory systems. During maintenance operations, fitters can download drawings, diagrams or manuals. As the wind turbine has its own e-mail address, rapid discussion of an arisen problem is possible with the Bonus maintenance section and the component producers.

For the staff on shore this means an improved monitoring, increasing the safe operation of the turbine.

For the customers, the new system offers the possibility of a rapid and precise control of the energy production from single turbines and from the wind farm as a whole.

The computer for the network monitoring system is connected to the wind turbine control system. This can also be controlled independently in the event of a possible computer breakdown.

The installed computer is not a normal PC, as we know it from home or the office. It is a robust and highly reliable industrial model.

A traditional monitoring system is installed, functioning as a back-up system independent of the network based system. This reserve monitoring system transmits and receives information via a fibreoptical cable using a communication card.

## COMPONENT ANALYSIS

The wind turbine computer is also used for component analysis based on data on vibration levels in the different components and areas of the wind turbine.

A thorough analysis of vibration data can be used for better planning of maintenance and to reveal possible future component failure. The overall result is increased operational security and a more efficient maintenance.

The component analysis is based on a number of so called "intelligent accelerometers" fixed to the main components: The main bearing, the main gear and the generator.

Data acquisition and processing is partly carried out in the intelligent accelerometers themselves, and therefore large data quantities can be handled without overloading the internal wind turbine communication system. The processed data are regularly exported from the meters to the monitoring PC.

An individual accelerometer has vibration sensors, a RS485 communication unit, a digital signal processor, temperature measuring unit, relay- and analog outputs. The accelerometer can analyse several things including the FFT (Fast Fourier Transformation) auto spectrum, patterns of displacement and time series.

A wind turbine is usually equipped with a single 2-channel and two 1-channel accelerometers. However, up to 32 units can be connected to the same network.

The individual accelerometers can be reprogrammed from the installed computer and from the Bonus factory or from the customer's computer. By reprogramming, interesting or new vibration patterns can be more thoroughly analysed by the accelerometer.

The most important data for the monitoring system is the averaged FFT spectrum from each individual accelerometer. This enables identification of each frequency in the bearings and in the gearwheel mesh

It allows a technician to detect future bearing and gearwheel failures, providing the possibility of choosing appropriate preventive maintenance.

Other types of analysis can be chosen, should a fault need to be more detailed.

The output signals from the accelerometers can be analysed by industrial analysis programs, such as those made by the company Brüel & Kjær, or they can be processed by Bonus´ own software system.

The analysis control system is especially valuable for offshore wind turbines. However, it is also fitted as standard equipment in our 2MW land based turbines.

## OTHER MODIFICATIONS

Apart from small alterations carried out on any individual wind turbine, there is often a need for specific site-relevant project modifications:

*Tower modifications enabling the installation of high frequency electrical equipment

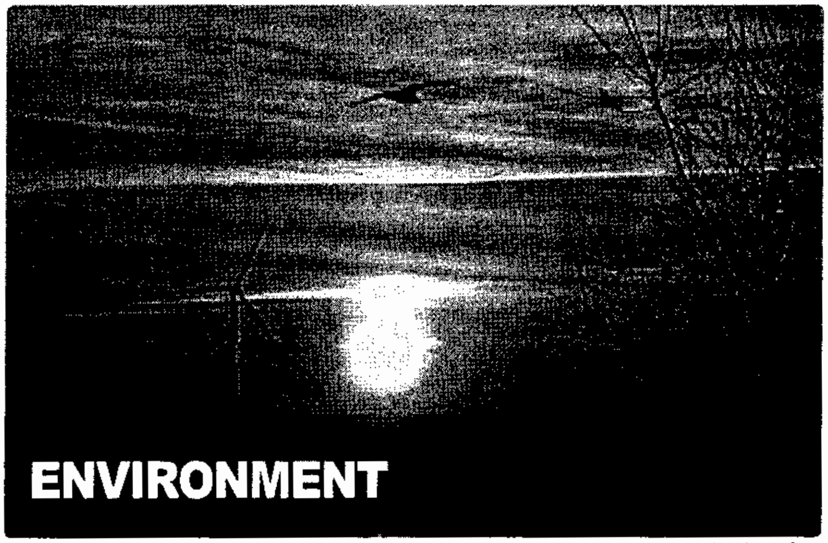
*High capacity UPS (Uninterruptible Power Supply) back-up systems

*Survival-room for fitters on sites with a high risk of being stranded as a result of sudden weather deterioration

*Automatic front illumination

*Aircraft warning lights

*Special boarding arrangements




*Bonus 2 MW as seen from the Amagerværket, the utility power plant nearby* Jens Hansen, Seas

**In connection with the project's public authorization, a large-scale environmental evaluation was presented**

The evaluation was conducted by the Københavns Energi- og Miljøkontor in cooperation with the company EMU Energy and Environmental Investigation. This report can be downloaded from the wind turbine cooperative's web site at "www.middelgrunden.dk". The main conclusions are presented below in a shortened and revised form.

## THE MIDDELGRUNDEN

The Middelgrunden is a natural reef lying roughly north-south, bounded by the navigation channels of the Kongdybet to the west and the Hollænderdybet to the east.

The greater part of the area is shallow with depths under six metres. At the wind turbine sites the water has a depth of app. three-four metres. Such low waters are important as habitat areas for aquatic plants and animals. As the wind farm has an environmental impact on a total area of 3.7 km², special considerations must be observed.

For more than a century the Middelgrunden was used as a dumping ground for rubble, old building materials and polluted harbour mud. The total effect of this dumping is in reality unknown, and all dumping activity ceased in 1984.

In 1997, the Geoteknisk Institut carried out seabed core sampling on possible wind turbine sites. These drillings indicated that about 5.5 % of the samples were polluted and 18% of them were highly polluted with one or more heavy metals. The concentration of mercury was 3.6-10.9 %.

## EXCAVATION

During excavation it is unavoidable that a certain portion of the excavated material will be spilled and mixed with the environment, thus entering the food chains. However, it was possible to limit the spillage to less than 5 % during the construction of the Øresund Bridge from Denmark to Sweden

Traditional hydraulic digging techniques were used during this project, where care was taken to disturb the water as little as possible. The same procedure is used on the Middelgrunden.

The water current will either take the spillage to a large bassin to the north, or lead it to be diluted when merging with strong currents in the area south of the project

There is a continued natural erosion from the Middelgrunden, with heavy metals being washed away. The construction of the wind farm will only have a minor short term effect.

Finally it should be noticed that excavation also creates muddy water with a short term shadow effect for plants and animals. This shadow effect is calculated to have a radius of 100 metres from each foundation i.e. 4 % of the Middelgrunden's total area. As excavation on each site will be limited to a few days, the shadow effect will have little relevance for flora and fauna.

## PLANTS

50 % of the seabed on the Middelgrunden is covered with vegetation, mainly eelgrass acting as an important spawning ground for fish and their fry. Also, plant eating birds eat eelgrass, and the grass roots contribute to stabilizing the seabed from the eroding effects of waves and water currents.

Ten of the twenty wind turbines stand in areas with vegetation, and when including cable trenches roughly 7.500 m² of the eelgrass area is effected, which is 0.004 % of the total area. Additionally 700 m² of plants are covered during foundation scraping. However, eelgrass reproduce by their roots, and in connection with dumping it has shown a propensity to return after a few years.

## WATER LIVING ANIMALS AND FISH

The cod lives around the borders of the Middelgrunden, but on the reef itself mostly eels and blue mussels are to be found. Mussels live on the large banks covering about 10-15% of the area. They filter water, are of great importance for water quality and provide food for eiders and diving ducks.

Foundations act as artificial reefs, and many animals and plants settle on them, including blue mussels and seaweed. It is therefore considered that the Middelgrunden will experience an additional influx of species - resulting in improved feeding possibilities in the area.

This supposition is confirmed by investigations near the foundations of the offshore wind farm at Vindeby, erected by Bonus in 1991. In comparison with a nearby reference area, the wind farm site has increased stocks of crayfish and fish including the cod.

It is reasonable to expect that a similar process will follow on the Middelgrunden, and that within some years additional plants and animals will migrate to the area as a result of the wind turbine project.

Restrictions on sailing and fishing will only be maintained to a limited extent. Seamarkers will give warnings and prevent fishing, anchoring and installing fishing stakes within 200 metres from buried sea-cables.

It has been discussed whether the fish are effected by the magnetic field emitted from the sea-cables. The conclusion is that the earth's own magnetic field will be the dominating factor within a short distance from the cables.

## BIRDS

The Middelgrunden is near the Copenhagen harbour, and usual port activity results in a relatively decreased bird life apart from seagulls.

No birds actually nest on the Middelgrunden, but they use it as a resting area. The relatively shallow waters with blue mussels and eelgrass also provide food for seabirds such as eiders, mute swans, red breasted mergansers and a variety of ducks.

The Middelgrunden is overflown every morning and evening by birds flying between Copenhagen and the small flat island of Saltholm. In the spring and autumn smaller flocks of migrating eiders alight on the area as a resting ground.

Light and sound disturbances in connection with turbine erection can probably disturb the birds, however as the numbers concerned are relatively minor, it is expected that they will move away only during constructional operations.

Noise levels during foundation construction in the dry dock and loading activities on the quay will correspond to noise from a medium-size construction site. Therefore, no additional problems are expected.

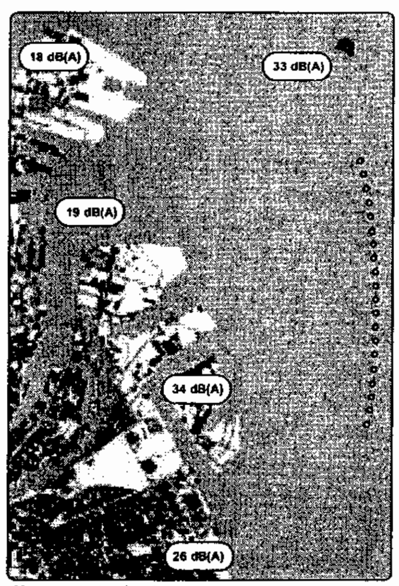
Noise during wind turbine operation and their sheer presence will disturb the birds, but in the worst case they will just keep their distance. However, increased food possibilities resulting from an increased growth of seaweed and blue mussels on the foundations could result in an increased bird life.

It is often asserted that birds fly into wind turbines and get killed. However, many investigations has shown that risks of collision are minimal. Wind turbines can both be seen and heard, and in addition large wind turbines present a lower collision risk than smaller models.

## WATER CURRENT

The Øresund connects the Kattegat and the Baltic and the water current has great significance for fish spawning. A comprehensive computer model was used to evaluate the water currents in connection with the Øresund Bridge construction. This model also applies to the Middelgrunden project and shows the highest possible reduction in the flow of water current to be 0.005 % resulting from the construction.

## NOISE



*Noise emission levels*

Wind turbine noise emissions have been calculated for a number of nearby positions, and the noise level limits of 40 dB(A) for habited and recreational areas and 45 dB(A) out in the open country are never exceeded. The above map shows the noise levels at the locations closest to the site.

## ACCIDENTS

Wind turbines on the Middelgrunden poses risk of collision with ships. Already before the erection of wind turbines, sea charts over the area warned ships against passage due to the risk of grounding. Therefore, only smaller vessels sail into the area.

Statistical calculations predict that there is a possibility of ship impact on 1-2 wind turbines during the next 20 years. On the other hand, the presence of the wind turbines will contribute to reducing the risk of possible groundings in the area. Carl Bro A/S has made statistical calculations indicating that these two opposite tendencies will result in a total lower risk of accidents.

The largest environmental danger during ship collision and grounding is the risk of oil leakage from ships and wind turbines. Calculations suggest that oil leakages will occur in about 4 % of possible future collisions with a total spillage of about 150 $m^3$ pr. collision.

An individual wind turbine contains 1.5 tonnes of oil in sealed systems, and the risk of oil-spills from collision or from operational activity is therefore minimal.

## DISASSEMBLY

The foundation can be removed by lifting or by cutting it away from the base plate. Cables can be raised during the same process. Apart from blades and insulation, all material can be recycled, for example as ballast in road buildings or in new concrete constructions. None of these processes present special environmental difficulties.

## SAVED POLLUTION

An annual electricity production of 80 million kWh from the Middelgrunden wind fram will save the environment from the following emissions.

| | |
|---|---|
| Sulphur dioxide ..... | 250 tonnes |
| Nitric oxides .......... | 200 tonnes |
| Carbon dioxide ... | 60.000 tonnes |
| Slag and fly ash .... | 4.500 tonnes |

# Statement at public hearing held on December 7, 2004 at Mattacheese Middle School by the U.S. Army Corps of Engineers and the Massachusetts Environmental Policy Act Office on a proposal by Cape Wind Associates to build 130 wind turbines in Nantucket Sound.

Douglas Giuffre
Economist, Beacon Hill Institute at Suffolk University, Boston, MA.

## *Public Health Impacts and Economic Costs from Power Plant Emissions*

The Army Corps Draft Environmental Impact Statement (DEIS) concludes that the Cape Wind Project could have a "cumulative beneficial effect on public health, and result in a related reduction in the costs of adverse health impacts from existing power plant emissions...The yearly monetary savings associated with these reductions in adverse public health impacts is estimated at approximately $53 million dollars."

This estimation is based on a flawed extrapolation from the findings of a Harvard Public Health study, which focused on improving emissions at two of the nation's worst polluting power stations. ***We believe this extrapolated estimate largely overstates the annual monetary savings; we find the savings to be in the range of $7-$20 million and declining over time.*** The difference arises from the assumption, made originally by Cape Clean Air and reproduced by the Army Corps, that the wind park will offset production at either the Salem Harbor or Brayton Point power stations. This assumption, which we believe to be erroneous, is not supported by any evidence.

*The Army Corps' Estimates*

Below is a reproduction of Army Corps' Table 5.16-4. The table reports the estimated amount of pollutant reductions attributable to the wind park, assuming the wind park output offset production of a) the marginal producer in New England, based on an ISO-NE marginal emissions analysis, or b) each of the selected power plants on a one-for-one basis.

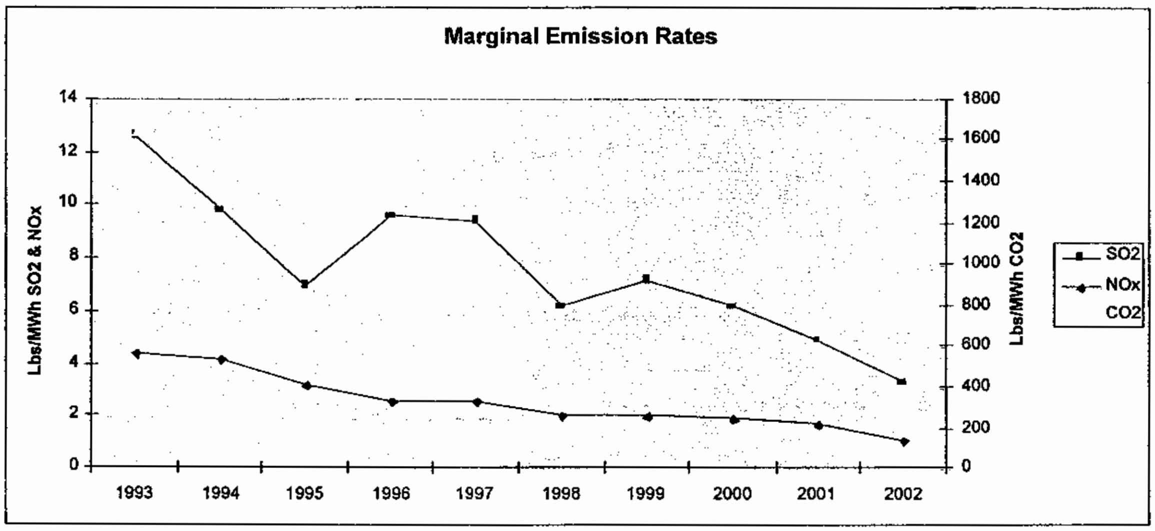
**Table 5.16-4 – Pollutant Emission Reductions using Wind Park Average Contributions (Tons/Year)**

| Reference | CO2 | SO2 | NOX | PM | CO | VOC |
|---|---|---|---|---|---|---|
| ISO NE Marginal Emission Rates | 1,108,039 | 4,606 | 1,415 | N/A | N/A | N/A |
| Salem Harbor | N/A | 9,800 | 2,600 | 11 | N/A | N/A |
| Brayton Point | N/A | 11,200 | 2,460 | 68 | N/A | N/A |

be offset by Cape Wind Energy, are 2-3 times lower than both Salem and Brayton Point.

2. Where the ISO-NE marginal emission rates are used, the Army Corps relies on outdated information.
    a. The ISO-NE numbers are based on 2000 data. Since this time, cleaner power sources have come online and marginal emission rates have continued their considerable downward trend.
    b. Updated numbers were available in January 2003 and again in December 2003. Figure 1 shows the continued downward trend in emissions as cleaner plants come online and older plants are retrofitted.

**Figure 1: Marginal Emission Rates, 1993-2002**




*Proposed Methodology*

Rather than rest an entire analysis on the assumption that Brayton Point and Salem will be offset, it is possible to use estimates of public health impacts per unit of emission ($/ton) derived by the Harvard Public Health studies. This allows for a monetary savings estimate to be derived using a reliable, published estimate yet based on the ISO-NE Marginal Emission analysis, a reliable simulation of the regional energy market.

Table 2 below illustrates the public health costs per unit of emission derived in "Development of a New Damage Function Model for Power Plants: Methodology and Applications" by Jonathan Levy et al. of the Harvard School of Public Health. The values have been updated to 2004 dollars using the Bureau of Labor Statistics Consumer Price Index.

*Conclusion*

While the assumptions and analysis of public health benefits attributable to the wind park seemingly rest on rigorous scientific studies, the overly simplistic extrapolation dramatically overstates the public benefits. Accurately quantifying these benefits is important in the context of a true cost-benefit test to which this project should be subjected, given the substantial public investment required for this project ($382 million by our estimation). The public deserves a better return on its massive investment and a better analysis of its benefits and costs.

**Statement at public hearing held on December 16, 2004 at MIT (Room 10-250) by the U.S. Army Corps of Engineers and the Massachusetts Environmental Policy Act Office on a proposal by Cape Wind Associates to build 130 wind turbines in Nantucket Sound.**

Dr. Jonathan Haughton
Department of Economics, Suffolk University, Boston, MA.

## *Economic Costs Exceed Economic Benefits for the Cape Wind Project*

Thank you. I wish to focus my remarks on just one point that poses – or should pose – an insurmountable obstacle to the wind farm project. Presidential Executive Order 12866 of September 30, 1993 states that "each agency shall ... propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs." The Draft EIS itself notes (p.2-2) that "the benefits which reasonably may be expected to accrue from the proposal must be balanced against its reasonably foreseeable detriments."

Although comments on costs and benefits are to be found scattered throughout the Draft EIS, the Army Corps does not, however, directly address the bottom-line question: "Are the social benefits of the project greater than the costs?" More importantly, it turns out that when one does, in fact, address this question, the answer turns out to be, "No": The benefits do not, in fact, measure up to the costs.

That they do not is made clear in a report submitted to the Army Corps on May 14 of this year by the Beacon Hill Institute at Suffolk University.[1] As the principal author of this report, I can state that, on the basis of the available facts, the wind farm project fails a cost-benefit test of the kind envisioned by the Presidential Executive Order. My interest in cost-benefit analysis is long-standing: I have taught the subject at Harvard University, Suffolk University and elsewhere, since 1987.

In our analysis we estimate the economic costs of the project to be 9.06 cents per kWh of electricity produced, very close to the figure of 9.00 cents reported in the Draft EIS (p.3-307). This is expensive for factory-gate electricity – on my most recent bill from N-Star I paid 6.32 cents for the generation costs of the electricity I used.

[1] Jonathan Haughton, Douglas Guiffre, David G. Tuerck and John Barrett. *An Economic Analysis of a Wind Farm in Nantucket Sound.* Beacon Hill Institute at Suffolk University, Boston. April 2004.

| **Table 1:** ***Economic Costs and Benefits of the Nantucket Sound Wind Farm Project*** | | | |
|---|---|---|---|
| | **Net Present Value (at 10%)** | | **Cents/kWh** |
| | **Mean** | **90% confidence interval** | |
| | | *($ millions)* | |
| **Benefits** | **744** | **638-859** | **7.06** |
| *Of which:* | | | |
| Fuel saved | 522 | 455 – 597 | 4.95 |
| Capital and operating costs saved | 104 | 85 – 122 | 0.98 |
| Emissions reduced | 108 | 55 – 176 | 1.02 |
| Greater energy independence | 11 | 3 – 21 | 0.10 |
| **Costs** | **952** | **888 – 1,035** | **9.06** |
| *Of which:* | | | |
| Project itself | 888 | 824 – 969 | 8.45 |
| Grid integration | 26 | 23 – 28 | 0.24 |
| Environmental effects (using royalty rates) | 39 | 35 – 44 | 0.37 |
| **Benefits - Costs** | **(209)** | **(333) – (83)** | **(1.99)** |
| Costs using expected property value | **(1,520)** | (1,647) – (1,392) | |
| Costs using willingness to pay measure | **(173)** | (300) – (46) | |

*Note:* Totals may not add exactly, due to rounding errors.
Based on 10,000 drawings from underlying distributions of the variables determining costs and benefits.

| **Table 2:** ***Reconciling Private and Economic Returns*** | | |
|---|---|---|
| | *Cents/kWh* | *PV, $ millions* |
| **Private return on equity (from Table 3)** | **0.29** | **30** |
| **Plus external benefits:** | | |
| + Capital and operating expenditures saved | 0.99 | 104 |
| + Value of emissions abated | 1.03 | 108 |
| + Value of greater energy independence | 0.10 | 11 |
| + Taxes paid to Federal, State and Local governments, and royalties | 0.39 | 41 |
| **Less external costs:** | | |
| – Cost of integrating wind power with New England grid | 0.24 | 26 |
| – Environmental/aesthetic costs | 0.37 | 39 |
| – Federal production tax credit | 0.94 | 98 |
| – Massachusetts green credits | 2.55 | 267 |
| – Accelerated depreciation for tax purposes | 0.55 | 58 |
| **And technical adjustments** | | |
| + For value of output (economic valuation > market valuation)* | 0.28 | 29 |
| – For loan effect (developer can use optimal loan financing)** | 0.41 | 43 |
| **= Net Economic Benefits (from Table 1; Benefits – Costs)** | **(1.99)** | **(209)** |
| **Memo items:** | | |
| Actual subsidy (net of taxes) | 3.65 | 382 |
| Optimal subsidy | 2.56 | 268 |
| **Therefore: excess subsidy** | **1.09** | **114** |

*Notes:* * The market valuation measures what Cape Wind receives from selling the electricity from the project; the economic valuation measures this as the value of energy saved (which is slightly higher than the market valuation). ** The developer has recourse to loan financing, which raises the private return on equity since the interest rate on loans is lower than the discount rate of 10%.

# Results of the electromagnetic investigations and assessments of marine radar, communications and positioning systems undertaken at the North Hoyle wind farm by QinetiQ and the Maritime and Coastguard Agency

Martin Howard and Colin Brown

QINETIQ/03/00297/1.1
MCA MNA 53/10/366
22 November 2004

Requests for further information should be sought from:

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The Maritime and Coastguard Agency
Spring Place
105 Commercial Road
Southampton
Hampshire
SO15 1EG

# Administration Page

| **Customer Information** | |
|---|---|
| For QinetiQ | For MCA |
| Project title<br>Electromagnetic investigations at the North Hoyle wind farm | Project title<br>The effects of offshore wind farms on marine radar, navigation and communication systems |
| Customer organisation<br>NPower Renewables Ltd. | Customer organisation<br>Department for Transport, Shipping Policy 2 |
| Customer contact<br>Stephen Bolton | Customer contact<br>John Mairs |
| Contract number<br>PRO032481 | Contract number<br>MNA 53/10/366 |
| Date due<br>22 November 2004 | Date due<br>22 November 2004 |

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| Date of issue 22 November 2004 | Date of issue 22 November 2004 |

**Record of changes**

| Issue | Date | Detail of Changes |
|---|---|---|
| 1.0 | 29 September | First release |
| 1.1 | 22 November 2004 | Minor amendments |

# Executive Summary

**Overview**

The Maritime and Coastguard Agency (MCA) has responsibility, on behalf of the Department for Transport of the UK Government, for the safety of navigation under the International Convention for the Safety of Life at Sea (SOLAS), for the direction and co-ordination of search and rescue operations and for the prevention of marine pollution.

In this context MCA has been consulted by the Department for Transport, of which it is an executive agency, and the Department of Trade and Industry's Offshore Renewables Consents Unit with respect to assessing all foreseeable marine safety risks associated with applications made by wind farm developers.

Since no large-scale off-shore wind farms existed in the United Kingdom until the North Hoyle site was developed, investigation into their potential effect on marine radar, communications and navigation systems was necessarily limited to desk top and laboratory research. The North Hoyle development therefore presented an opportunity for QinetiQ and MCA to carry out experimental field tests for the first time in the United Kingdom, the results of which would be used to inform the offshore wind farm consents process and those whose operations could be affected by resulting developments. MCA's participation in this research was funded by the Department for Transport's Shipping Policy Division.

**MCA trials**

MCA's programme was intended to assess the effect of the wind farm structures on marine systems in operational scenarios. The trials assessed all practical communications systems used at sea and with links to shore stations, shipborne and shore-based radar, position fixing systems, and the Automatic Identification System (AIS). The tests also included basic navigational equipment such as magnetic compasses.

The effects on the majority of systems tested by the MCA were found not to be significant enough to affect navigational efficiency or safety, and an on-going collection of data on such systems is expected prove these conclusions. This will be achieved by further trials, where seen to be necessary and through the collation of data observed by mariners.

Some reported effects, such as those on short range radio devices, will be further investigated as will some scenarios which could not be assessed during the trials period, such as helicopter search and rescue operations within wind farms.

The only significant cause for concern found by the MCA during the trials was the effect of wind farm structures on shipborne and shorebased radar systems. It was determined that the large vertical extent of the wind turbine generators returned radar responses strong enough to produce interfering side lobe, multiple and reflected echoes. While reducing receiver amplification (gain) would enable individual turbines to be clearly identified from the side lobes - and hence limit the potential of collisions with them - its effect would also be to reduce the amplitude of other received signals such that small vessels, buoys, etc., might not be detectable within or close to the wind farm. Mariners will require guidence on these potential effects. Bearing discrimination was also reduced by the magnitude of the response and hence the cross range size of displayed echoes. If on passage close to a wind farm boundary or within the wind farm itself, this could in some circumstances affect a vessel's ability to fully comply with the International Regulations for the Prevention of Collisions at Sea. For full compliance, mariners will need to pay particular attention to the determination of a safe speed and to assessing risk of collision when passing near or through wind farms, particularly in restricted visibility. The cited Regulations are contained in Appendix C of which the relevant sections are Rule 6(b) (ii) (iii) (iv) and (v), Rule 7 (b) and (c), Rule 19 (a) (b) (c) and (d). It was also found that the performance of a vessel's automatic radar plotting aid (ARPA), referred to in Rule 7 (b) ,could be affected when tracking targets in or near the wind farm.

With respect to the multiple and reflected echoes produced when wind farm structures lie between the observing radar and a relatively high sided vessel, gain reduction will have similar effects to those described above. If, as in the trial undertaken, a shore or platform based radar is intended to detect and track traffic in port approaches, Vessel Traffic Services (VTS) or in the proximity of off-shore oil or gas installations, the effects could be significant.

**QinetiQ trials**

The QinetiQ trials were designed to test the theoretical results calculated in previous work [1]. The previous work had calculated the expected effects of the wind turbines at the North Hoyle wind farm on marine communications, GPS and radar systems. In this report the experimental tests carried out to validate the theoretical results [1] are described. This work has been funded by NPower Renewables Ltd.

Four trials, covering the areas of GPS, VHF communications and radar tracking and radar clutter were performed by QinetiQ.

The QinetiQ GPS trial involved traversing previously defined courses through and around the wind farm. Along each course, the number of satellites visible to two different GPS systems (a Garmin 152 and a Garmin GPSIII) and the position of the ship were recorded. Our results show that on average between 8 and 11 satellites were visible at any one time providing accurate positioning to within 5 metres.

The effect of wind turbines on VHF communications was investigated by QinetiQ using a hand-held VHF transceiver that was run in series with an adjustable attenuator. A link margin of 1 dB was achieved in free-space (away from any turbines). This required an attenuation of 16dB to be added to the transceiver.

To explore the shadow region behind the wind turbines, four link margins, 2dB, 3dB, 4dB and 5dB were used. These link margins correspond to a total attenuation of 15dB, 14dB, 13dB and 12dB added to the transceiver. The closest approach to turbine 21 was 500 metres and approximately 5m behind turbine 26. As expected the depth of shadow was greater when closer to a turbine. When behind turbine 21 the shadow was found to be approximately 2dB to 3dB lower than the attenuation needed to give a 1dB link margin in free space. For turbine 26 the shadow was deeper due to the closer proximity of the VHF system. It was found that behind turbine 26 the depth of shadow was approximately 10dB below the link margin in free space. The shadow depths are shallower than predicted theoretically confirming the worst case expectations of the theoretical work.

The QinetiQ radar shadowing trials provided very little evidence that shadowing of targets would present any significant problems. In particular the shadowing observed was, like the VHF trials, less than predicted in the theoretical study. Clutter in the radar display due to the presence of wind turbines was found to be quite considerable. Both ring-around and false plots were observed (referred to by mariners as side-lobe, multiple and reflected echoes). The observed problems could be suppressed successfully by using the gain and range settings of the radar. However, this may have the unwanted side-effect of no longer being able to detect some small targets.

**Conclusions**

The general findings were as follows:

i Global Positioning System (GPS)

No problems with basic GPS reception or positional accuracy were reported during the trials.

ii Magnetic compasses

The wind farm generators and their cabling, interturbine and onshore, did not cause any compass deviation during the MCA trials. As with any ferrous metal structure, however, caution should be exercised when using magnetic compasses close to turbine towers.

iii Loran C

Although a position could not be obtained using Loran C in the wind farm area, the available signals were received without apparent degradation.

iv Helicopter radar and communications systems

These trials were not carried out due to helicopter call-outs to emergencies on the trial days. The emergency services are keen that they should be undertaken when convenient. MCA will co-operate with RAF Valley and other emergency services to ensure that this is done.

v VHF and other communications

The wind farm structures had no noticeable effects on any voice communications system, vessel to vessel or vessel to shore station. These included shipborne, shorebased and hand held VHF transceivers and mobile telephones. Digital selective calling (DSC) was also satisfactorily tested. The VHF Direction Finding equipment carried in the lifeboats did not function correctly when very close to turbines (within about 50 metres) and the BHP telemetry or short range radio link to and from its deployed RIB (rigid inflatable boat) was similarly reported to suffer interruptions.

vi The Automatic Identification System (AIS) carried aboard MV "Norbay" and monitored by HM Coastguard MRSC Liverpool was fully operational.

vii Small Vessel radar performance.

1. The wind turbine generators (WTG) produced blind and shadow areas in which other turbines and vessels could not be detected unless the observing vessel was moving.
2. Detection of targets within the wind farm was also reduced by the cross and down-range responses from the WTGs which limited range and bearing discrimination.
3. The large displayed echoes of WTGs were due to the vertical extent of the turbine structures.
4. These returned strong responses from sectors of the main beam outside the half power (-3dB) points and the side lobes outside 10° from the main beam.
5. Although such spurious echo effects can be limited to some extent by reducing receiver amplification (gain) this will also reduce the amplification of other targets, perhaps below their display threshold levels.
6. Sea and rain clutter will present further difficulties to target detection within and close to wind farms. Weather conditions at the time of the trials were such that these effects could not be examined.

viii Shore based radar performance

1. Short range performance (less than 6 nm)
   When a small shore based radar was sited such that the height of its antenna was about six metres above sea level, its performance with respect to small vessels was similar to that of the vessel-mounted systems in terms of range and bearing discrimination and target detection within the wind farm.

   When moved to a height of 200 metres above sea level there was an improvement in range discrimination.

   When the higher powered and narrower beam width BHP Billiton radar was used, at the same height, the visual detection of targets within, and beyond, the wind farm was again improved.
2. Larger vessel detection
   A larger vessel was easily detected within and beyond the wind farm. However, while it was broadside on to the direction of the shore radar, reflections from the turbines produced strong multiple echoes. At an oblique aspect to the radar, multiple echoes did not occur, but some reflected echoes were observed.
3. Long range radar (more than 12 nm)
   When the wind farm was observed at long range by the Mersey docks and Harbour Board radar the vessel was easily detected and tracked

ix Radar and ARPA carried on larger vessels

As with small vessel radars, range and bearing discrimination were affected by the response from the WTGs. Definition was less on S band radar than on X band. Numerous spurious echoes from side lobes and reflections were reported by MV "Norbay" starting at a range of about 1.5 nm. The ship's ARPA had difficulty tracking a target vessel within the wind farm due to target swop to the stronger response. This substantiated a similar report with respect to the BHP Billiton radar's own tracking system

x Non type-tested radar, communications and navigational equipment

The effects on such systems will be similar to those tested during the trials but will vary individually with respect to transmitted power, antenna performance, radar beam width, etc. The Royal Yachting Association is assisting MCA by providing ongoing information through the experiences of its membership.

With the exception of those noted in the next paragraph, most of the effects of offshore wind farm structures on the practical operation of marine radar, communications and navigation systems are not anticipated to significantly compromise marine navigation or safety. Where questions are raised about specific systems during the on-going collection of data they will, when possible, continue to be monitored and assessed.

There are however concerns about the use of both shipborne and shorebased radar as an effective aid to both vessel and mark detection and, consequently, for ship-to-ship collision avoidance in the proximity of wind farms. Wind farm structures generally have high vertical extents and therefore will return very strong responses when observing radars are close. The magnitude of such responses will vary according to transmitted radar power and proximity to the structures but can prevent both the visual detection of targets and the effective operation of automatic radar plotting aids (ARPA). These effects can be mitigated by vessels keeping well clear of wind farms in open water or, where navigation is restricted, keeping the wind farm boundaries at suitable distances from established traffic routes, port approaches, routing schemes, etc. Other technical solutions may be employed, particularly in port approaches.

For a particular wind farm these boundary distances should be determined in consultation with the Maritime and Coastguard Agency's Southampton HQ in conjunction with other stakeholders and included in the Environmental Statement submitted with the consent application. A Department of Trade and Industry (DTI) funded navigational risk assessment project is about to be undertaken. This will produce a methodology for assessing navigational risk - and marine risk in general - in and around offshore wind farms. It is intended to be used by government agencies for the assessment and, where appropriate, acceptance of offshore wind farm applications, and for the guidance of developers in the preparation of such applications. Included in this will be recommendations on suitable distances of wind farm boundaries from traffic routes . In the meantime, a set of recommendations based on domain theory, and taking into account the above effects, has been produced as a draft working template by MCA.

With respect to shorebased or offshore platform based systems, the careful siting of radar scanners in relation to traffic routes and wind farm configurations should enable any degrading effects to be minimised. Again, the location or relocation of required radar systems and their funding should be determined in consultation with the relevant organisations, these data included in the Environmental Statement, and submitted with the consent application.

Further work needs to be done, as for example identified in the report with respect to adverse weather conditions, helicopter search and rescue operations, short range radio systems, non type-tested systems, etc. These should be carried out as soon as is practical.

**Acknowledgements**

Many individuals, companies and organisations took part in these trials. In particular the QinetiQ and the Maritime and Coastguard Agency would like to record their appreciation for the contributions of the following:

Broken Hill Proprietary Billiton Ltd., its staff and the crew of the "Clwyd"

The Chamber of Shipping

Denbridge Marine Ltd

The Environment Agency and its staff in Buckley, North Wales.

Mersey Docks and Harbour Board

NPower Renewables and the crews of "Celtic Wind" and "Fast Cat"

The P & O Steamship Company Ltd., and the officers of M.V."Norbay"

The Trinity House Lighthouse Service and its staff.

The Royal National Lifeboat Institution with the shore and sea crews of the Rhyl and Hoylake lifeboats.

Paul Frost 2nd Mechanic of Rhyl Lifeboat Station for video camera work

The Royal Yachting Association

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# Abbreviations and Acronyms

| | |
|---|---|
| AIS | Automatic Identification Systems |
| ARPA | Automatic Radar Plotting Aid |
| BHP | Broken Hill Proprietary (Billiton) |
| CA | Cruising Association |
| dB | Decibels |
| DSC | Digital Selective Calling |
| DTI | Department of Trade and Industry |
| FTC | Fast Time Constant |
| GPS | Global Positioning System |
| IMO | International Maritime Organisation |
| ISPS | International Ship and Port Facility Security Code |
| KHz | Kilohertz (radio frequencies) |
| MCA | Maritime and Coastguard Agency |
| MDHB | Mersey Docks and Harbour Board |
| MRSC | Maritime Rescue Sub-Centre |
| MHz | Megahertz (radio frequencies) |
| MV | Motor Vessel |
| NFFO | National Federation of Fishermens Organisations |
| nm | Nautical mile (1852 metres) |
| OREI | Offshore Renewable Energy Installation |
| P & O | Peninsular and Orient ( Shipping Company) |
| RACON | Radar Beacon |
| RAF | Royal Air Force |
| RCS | Radar Cross Section |
| RIB | Rigid Inflatable Boat |
| RNLI | Royal National Lifeboat Institution |
| RPM | Revolutions Per Minute |
| RYA | Royal Yachting Association |
| SAR | Search and Rescue |
| SOLAS | Safety of Life at Sea |
| THLS | Trinity House Lighthouse Service |
| UHF | Ultra High Frequency |
| UK | United Kingdom |
| US | United States |
| $\mu s$ | Microseconds |
| VHF | Very High Frequency |
| VTS | Vessel Traffic Services |
| WTG | Wind Turbine Generator |



MCA and QinetiQ proprietary

# 1 Introduction

## 1.1 Background

Offshore wind farm installations are new to the United Kingdom and comparatively so to other countries' waters. The installations are large in area, and in the number and size of their structures. However, at the few sites where wind farms have been constructed, little detailed practical research on their effects on navigation and communications systems has been undertaken. Some relevant known research is listed in the reference section at the end of this report [5][6][7].

Experience with other types of offshore structure and the results of desktop studies indicated that offshore wind farm structures might have the potential to interfere with shipborne, shorebased and airborne radar, VHF communications and also - although with a lower probability - position fixing, guidance and Automatic Identification Systems (AIS).

Offshore wind farms, consented under Round 1 and proposed under Round 2, cover large areas of open water and hence present potential hazards to navigation. A number of them are considered to be close to or encroach into waters where there is a high density of shipping movements or be close to waters used by fishing vessels and recreational craft. Their positions are necessarily those which are exposed to weather conditions which could affect the navigation of vessels, particularly small craft. Their locations are, for technical reasons, in relatively shallow waters near shoals, and therefore in close proximity to restricted waters used by small craft and also shipping inshore gaining access to ports or to those waters providing a more sheltered passage required in inclement weather and sea conditions. Tidal streams of varying sets and rates pass through all wind farm sites. Some sites are within port limits and some lie within Vessel Traffic Services (VTS) operational limits.

Of necessity, when a vessel is within or close to a wind farm, mariners should be able to place similar reliance on marine navigation systems as in open sea areas, or they should be fully appraised of any induced errors or limitations which might be encountered. From the aspect of collision avoidance, vessels need to be able to detect other craft with which they might be in an encounter and to take appropriate avoiding action.

Port authorities and VTS operators require effective detection, identification and tracking of vessels navigating in their areas so as to be able to organise traffic or provide traffic information and navigational assistance services to vessels operating within port approaches or prescribed routing schemes to meet their statutory responsibilities in respect of the safety of navigation. The importance of effective detection and identification is further emphasised by the implementation of the International Ship and Port Facility Security (ISPS) Code from 1 July 2004.

Emergency services such as Royal National Lifeboat Institution (RNLI) vessels, HM Coastguard and RAF helicopters require the ability to rapidly detect and react to maritime casualties.

All of the foregoing require consistent and effective radio communications systems.

Failure of any radar, navigation or communication system could give rise to increased risks to safety or lead to marine casualties and insurance claims or reduce the effectiveness of emergency service operations. Incidents involving passenger vessels and those carrying dangerous and polluting cargoes could have serious consequences for the public and the environment, both at sea and ashore.

## 1.2 Objectives

The proposed research was intended to obtain scientific and practical operational data on various navigation and communications systems' performance within and in the vicinity of offshore wind farms. In particular, any degradation of the performance of systems was to be determined, quantified and, where considered necessary, cost effective solutions recommended. The offshore wind farm used in the investigation was the 30 turbine wind farm at North Hoyle, off the North Wales coast at Prestatyn. A map containing the wind farm is presented in Figure 1-1.

These data will be used to inform mariners, the shipping and ports industries, the General Lighthouse Authorities, the National Federation of Fishermen's Organisations, the emergency services, the Royal Yachting Association, wind farm developers and all other interested parties, of the extent of any system limitations, any consequent increased risks and, where necessary, recommendations as to how these should be mitigated.

This outcome may also be used to inform the consents process of offshore wind farm applications.

In addition to these aims, experiments were carried out to test the theoretical results from an earlier study [1]. This earlier study predicted the impact on marine radio systems by the North Hoyle wind farm.

In the theoretical study [1] it was found that wind turbines have very large radar cross-sections (RCS), which means that they will scatter a large proportion of any incident electromagnetic energy. In addition to this shadows will be cast behind the turbines looking from the direction of the transmitter.

The theoretical study suggested that small vessels within the North Hoyle wind farm would be detectable with marine radar (3GHz and 9GHz) if they were not in the shadow from a turbine. However, detection of the vessel could be compromised if it is very close and directly behind a turbine. The effect of the shadow at 3GHz was found to be much less severe than at 9GHz.

The impact on GPS was found to be minimal and any interference would very rarely cause any corruption to the GPS data. It was determined that unless a GPS receiver is within 70m (based on a signal-to-noise ratio of 15dB) of a wind turbine then any interference would be insignificant.

The theoretical study [1] also considered VHF communications. It concluded that due to the wavelength of the VHF systems any interference caused by wind turbines would be negligible.

Four different trials were designed to test the validity of the results from the theoretical study outlined above. The full technical details of these trials are presented in the trial plan[2].

## 1.3 Content

This report is separated into several sections that deal with the GPS, VHF communications and radar trials undertaken by QinetiQ and the MCA. In each section the experimental process is described and the results are presented in full. The structure to the report is as follows:

Section 2: QinetiQ GPS trials

Section 3: QinetiQ VHF communications

Section 4: MCA VHF communications

Section 5: QinetiQ Radar trials

Section 6: MCA Radar trials

Section 7: MCA marine navigation system trials


*Figure 1-1: The wind farm at North Hoyle*

# 2 QinetiQ GPS trials

## 2.1 Overview

The number of satellites visible to a GPS system bears a direct relation to the accuracy of the positioning. For the GPS system to work there must be line-of-site to at least four satellites. At any one time the GPS units can usually receive signals from up to twelve satellites. The more satellites that can be used in a positioning measurement, the more accurate the estimated position will be. The original theoretical study [1] demonstrated that it is unlikely that any electromagnetic interference will effect the normal operation of GPS system, unless the receiver is in very close proximity to a turbine tower.

The GPS trials consisted of piloting a launch along three predefined courses. Two control runs, away from the wind farm were also made. On each course the number of satellites used by the GPS receiver was recorded along with position. Two GPS systems were used, a Garmin GPSIII and a Garmin GPS152. The first is a typical hand-held GPS receiver and the second is typical of what might be found installed on small ships, launches and pleasure craft.

Full details of the experimental methods for the GPS trials can be found in the trial plan [2].

The antenna for the GPS152 was positioned on the cabin roof as illustrated in Figure 2-1. The hand-held GPSIII unit was positioned at the centre of the rear deck of the vessel.

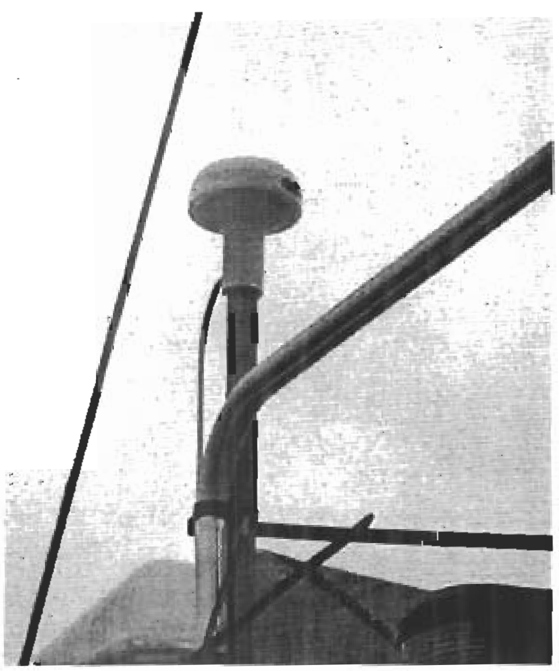


*Figure 2-1: The position of the antenna for the Garmin GPS152 unit*

## 2.2 Results

### 2.2.1 Control runs

Two control runs were made in order to determine the number of satellites visible when there were no possible obstructions to the line-of-site. The number of satellites locked with time is shown in Figure 2-2 for both the control runs.

Here we can see immediately that the visible number of satellites on each control run and for each GPS system is relatively stable in time. Furthermore, the total number of satellites visible is 9 for the GPSIII and 10 for the GPS152. This provides us with an expected number of satellites to work with when considering the different courses in and around the wind turbines. In addition to the expected number of satellites, we are also able to estimate the likely uncertainty in position estimation by the GPS units and compare these to the uncertainties provided when in the wind farm. In the control run, the recorded uncertainty in position was between 4m and 5m.

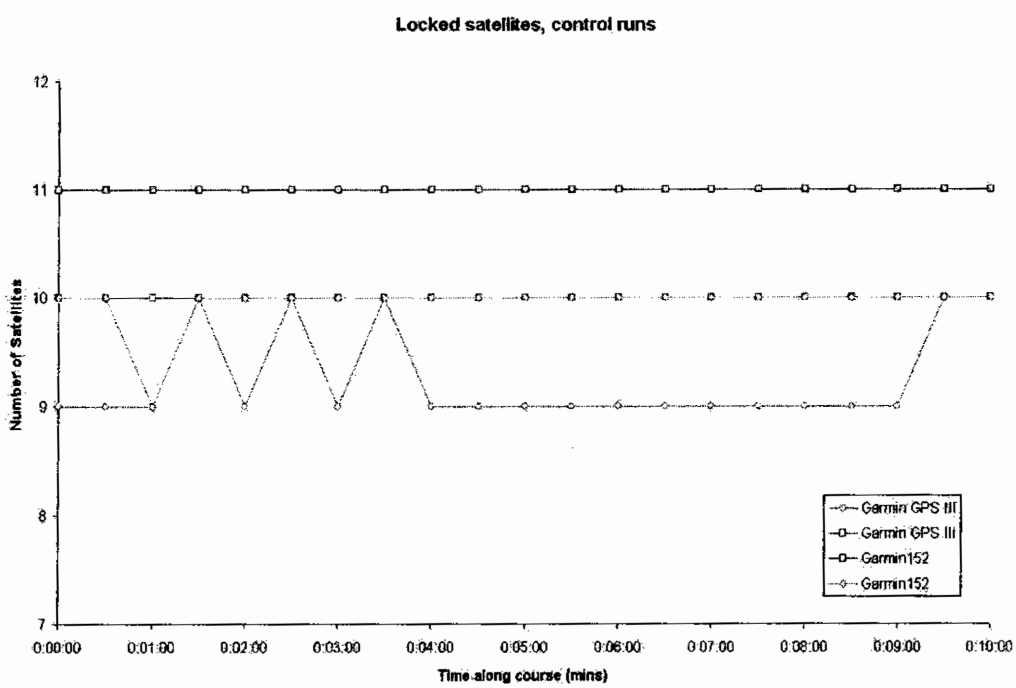



*Figure 2-2: Locked satellites on the two control runs*

In 2-3 and Figure 2-4 examples of the displays for the GPSIII and GPS152 units are shown. It can be seen in the figures that the number of satellites locked onto by the two GPS systems is eleven in each case. Furthermore, a twelfth satellite that is visible to the GPS152 unit.

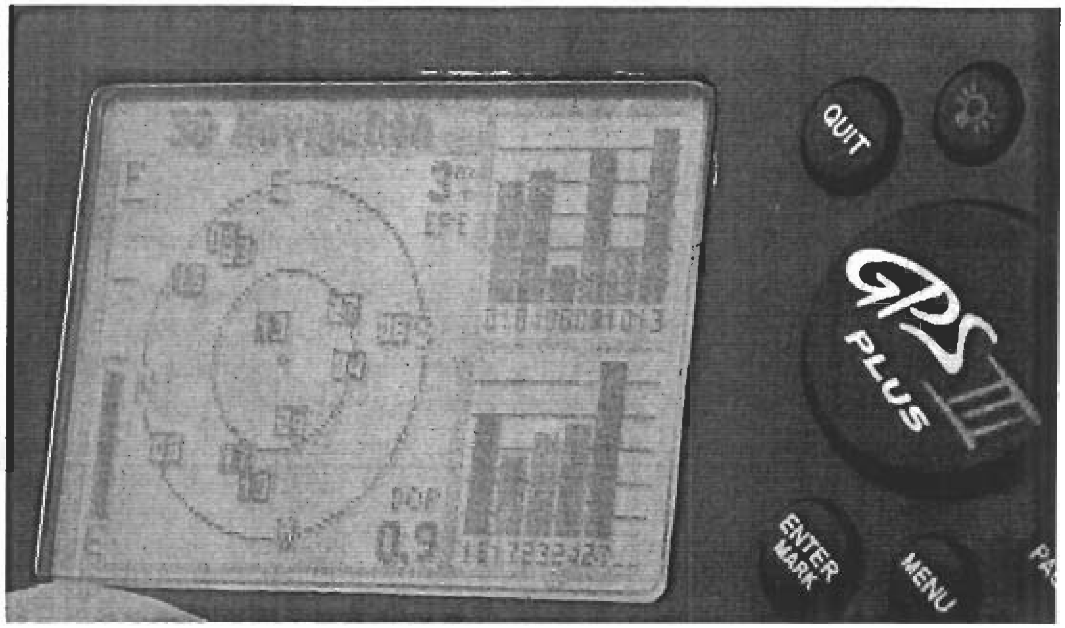



*Figure 2-3: The display from the Garmin GPSIII unit during a control run*

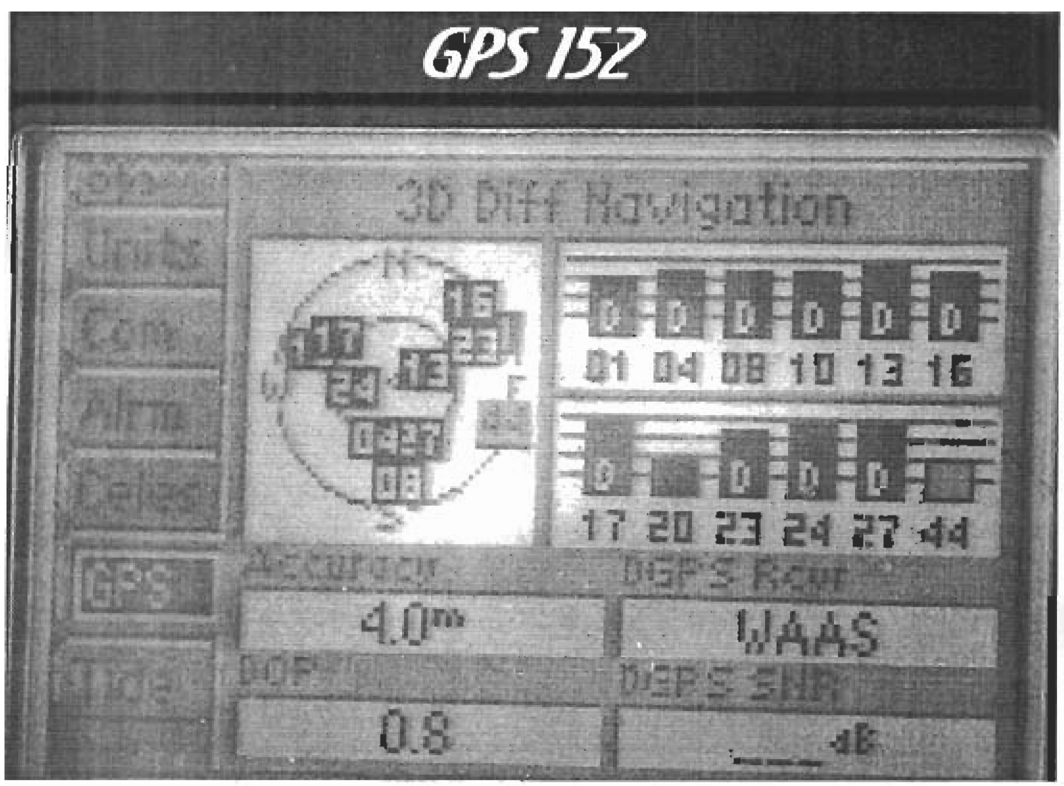



*Figure 2-4: The display from the Garmin GPS152 unit during a control run*

### 2.2.2 Trial courses

The track data recorded by both GPS units along the three predefined courses is plotted in Figure 2-5. The positions of the turbines are also indicated in the figure.

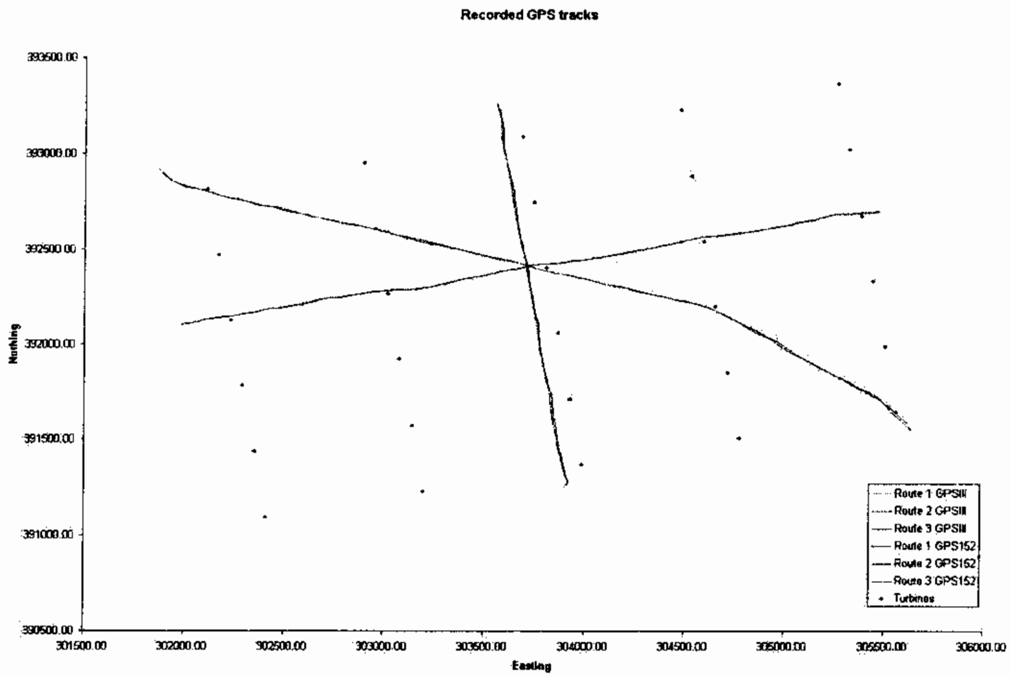



*Figure 2-5: The recorded GPS track data for the three routes used in the trial*

### 2.2.3 Course one

The first course is a path from the northern side of turbine 16 to turbine 20 (as described in [2]). The course runs in a direction parallel to the longest side of the wind farm as is shown by the green and brown lines in Figure 2-5.

In 2-6 we present the number of satellites locked onto by the GPS units with respect to time. It can be noted from the plot that for both the GPSIII and GPS152 the number of locked satellites is slightly less consistent than was seen in the control runs. However, for both GPS units between 8 and 10 satellites remains locked at all times providing an uncertainty in the estimated position of between 4m and 6m. It is important to note that for successful operation of a GPS unit, only four satellites are required. A greater number of satellites provide a greater accuracy in position.

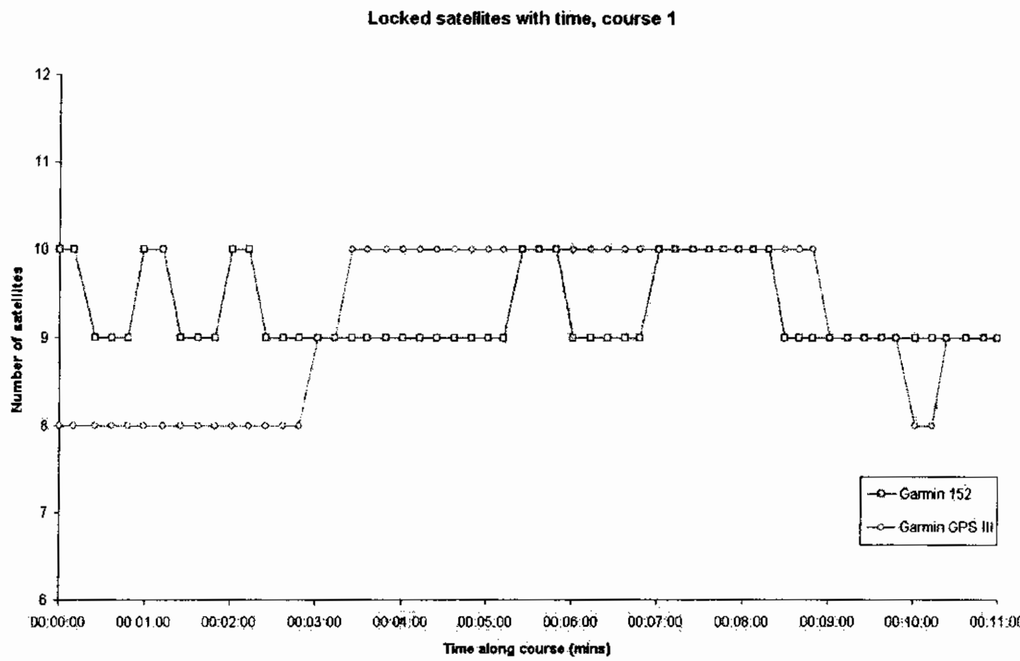



*Figure 2-6: The number of satellites locked onto by the GPS units along course 1*

### 2.2.4 Course two

The second course used to test the GPS systems ran parallel to the shortest side of the wind farm from the western side of turbine 3 to turbine 28 (see the blue and purple lines in Figure 2-5).

We found that on the course the number of satellites locked onto were 8 for the GPSIII and 10 for the GPS152. The uncertainty in position was recorded as 5m. It is interesting to note that the GPS152 appears to have a consistently higher number of satellites than the hand held GPSIII. However, this is likely to be a result of the elevated position of the GPS152 antenna (on the roof of the launch cabin). The hand held antenna was much lower on the boat and thus more susceptible to shadowing from objects other than the wind turbines. The results for the second course are presented in Figure 2-7.

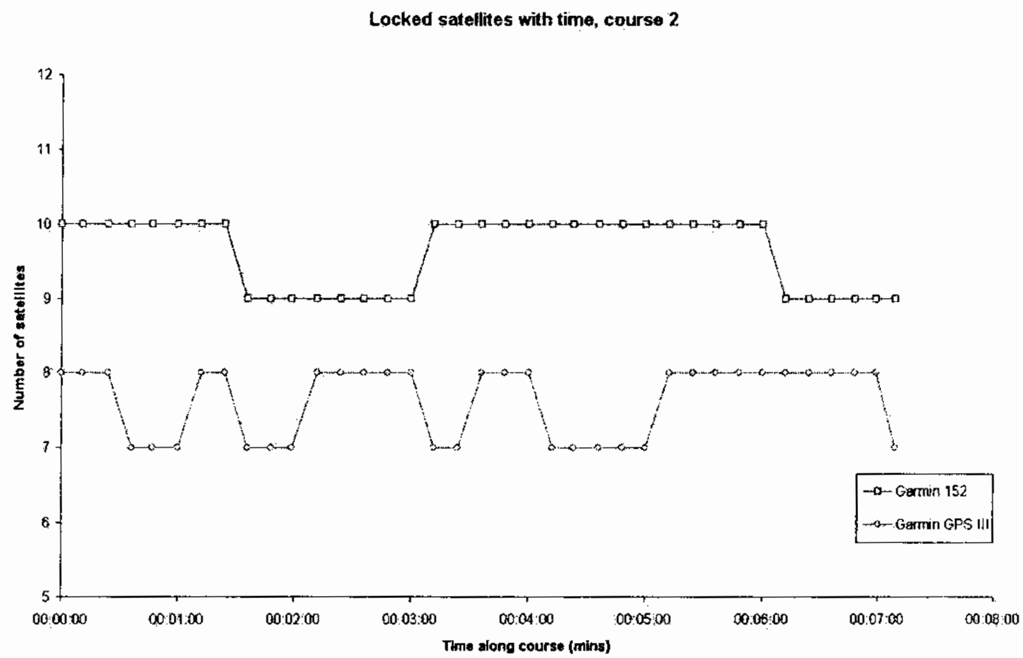



*Figure 2-7: The number of satellites locked onto by the GPS units along course 2*

### 2.2.5 Course three

The vessel was piloted diagonally through the wind farm from the south of turbine 5 to the south of turbine 26 and the data log of the course is shown in Figure 2-5 (red and light blue lines).

Here we find that there is very little variation in the number of locked satellites for either GPS system. The data is shown in Figure 2-8 and it can be noted that the GPSIII has 8 or 9 satellites locked at all times. The uncertainty in the positioning is around 4m. The GPS152 has 8 to 11 satellites locked and because of the variation in satellite number, the uncertainty in position was found to be much more variable, being between 3m and 5m. However, despite this overall operation of the GPS units was not affected adversely at any time.

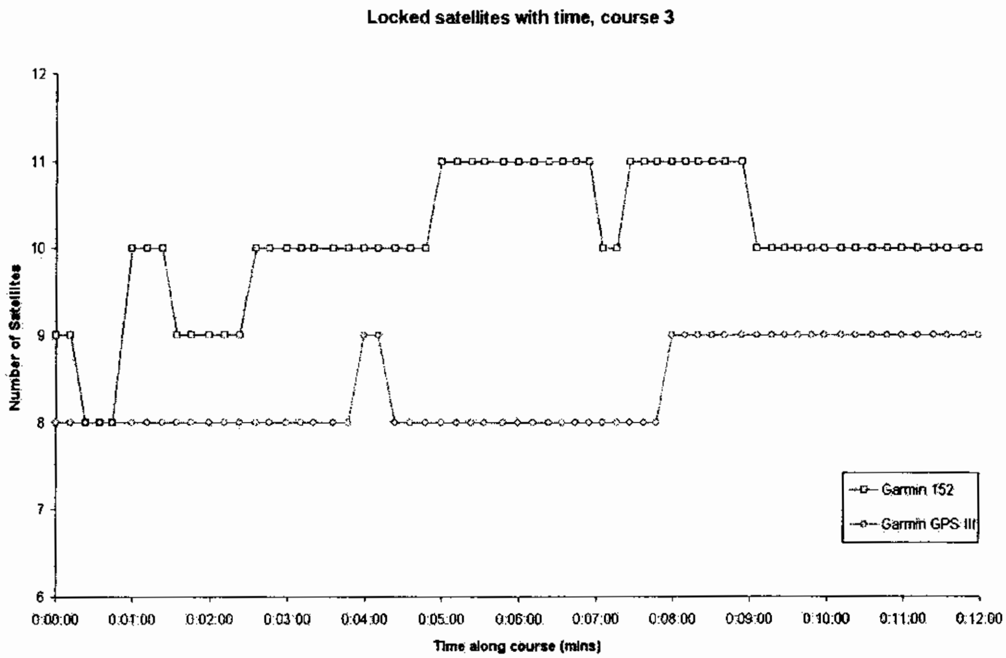



*Figure 2-8: The number of satellites locked onto by the GPS units along course 3*

### 2.2.6 Additional tests

In addition to the courses described above the GPS units were tested whilst the launch was stationary and adjacent to a turbine. Four turbines (numbers 7, 9, 13 and 17) within the wind farm were used in an attempt to shadow different parts of the sky.

We found that regardless of our proximity to a turbine the GPS units operated normally without any undue loss in the number of visible satellites. The results are summarised in Table 2-1. It should also be noted that in each case the estimated error in position with both the GPSIII and GPS152 was between 3 and 5m.

| | Number of satellites locked | |
|---|---|---|
| Turbine | GPS152 | GPSIII |
| 7 | 11 | 11 |
| 9 | 10 | 11 |
| 13 | 10 | 10 |
| 17 | 11 | 11 |

*Table 2-1: Summary of visible satellites when adjacent to a turbine*

## 2.3 Summary

The various experiments performed during the GPS trial showed that the wind turbines did not give rise to any loss in the number of locked satellites. The significant outcome of this is that the normal operation of the GPS system was never at risk of failure, due to interference from wind turbines.

The additional tests showed that even with a very close proximity of a turbine tower the GPS antenna, there were always enough satellites elsewhere in the sky to cover for any that might be shadowed by the turbine tower.

# 3 QinetiQ VHF communications

## 3.1 Overview

The use of VHF communications within the maritime community is wide spread. It is used for both ship-to-shore and ship-ship communication. It is essential that such communications are free from interference induced by intermediary structures since they are used in emergencies. The theoretical results have shown that the shadow at VHF frequencies behind a wind turbine tower is relatively shallow and should not adversely affect the normal operations of any VHF communication system. The VHF trial was designed to assess the depth of shadow behind wind turbine and compare the trial results with those expected theoretically.

The trial consisted of traversing a course that passed within 5m behind turbine 26 . A continuous communication to the receiver set up on the shore at Prestatyn was used. The track data along the course was recorded to provide an indication of when the vessel was in the turbine shadows, thus affecting the signal. The antenna and receiver set up at Prestatyn is shown in Figure 3-1. Link margins of 2dB, 3dB, 4dB and 5dB were employed to estimate the depth of shadow experienced. The link margin is the strength of the signal received above the noise level. In free space at a fixed range the link margin was found to be 17dB (i.e. the signal is 50 times stronger than the noise level). We added an attenuation of 16dB to reduce the link margin to 1dB above the noise level and this was used as the baseline for all the VHF tests.

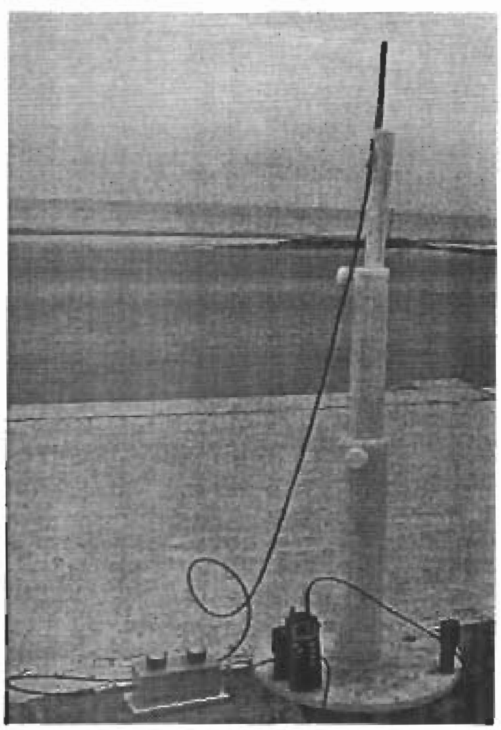


*Figure 3-1: The VHF antenna and receiver set up at Prestatyn*

## 3.2 Results

In free space, away from the wind turbines, to get the link margin of 1dB required an attenuation of 16dB to be added in series with the receiver antenna.

The results from all the different link margins are plotted together in Figure 3-2 and in Figure 3-3. The first of these figures shows the courses taken by the vessel when a 2dB and 3dB link margin was being used. In each case the uncertainty in our measurement is 1dB. On the graphs, the loss of signal is represented by the sudden drop in northing on the track. This "drop" shows the point at which the VHF signal was lost. The projection of the turbine shadows are shown as thick black lines.

In Figure 3-2 it can be seen that the shadows from turbines 26 and 21 have contributed to a loss in the VHF signal. It can also be noted in the figure that with a 2dB link margin there is a loss in the signal that occurs between the easting values of 301913m and 301942m. Similarly another loss, not attributable to any turbines exists around the easting value of 302075m. These are the result of interference from other sources, such as another broadcasts on the same VHF channel.

Turbine 21 is approximately 500m from the path of the launch. At this distance behind a wind turbine the shadow predicted is approximately 2dB (at 150MHz). Considering that the uncertainty in the link margins is of the order of 1dB, our experimental results are in very good agreement with the predictive work undertaken previously [1].

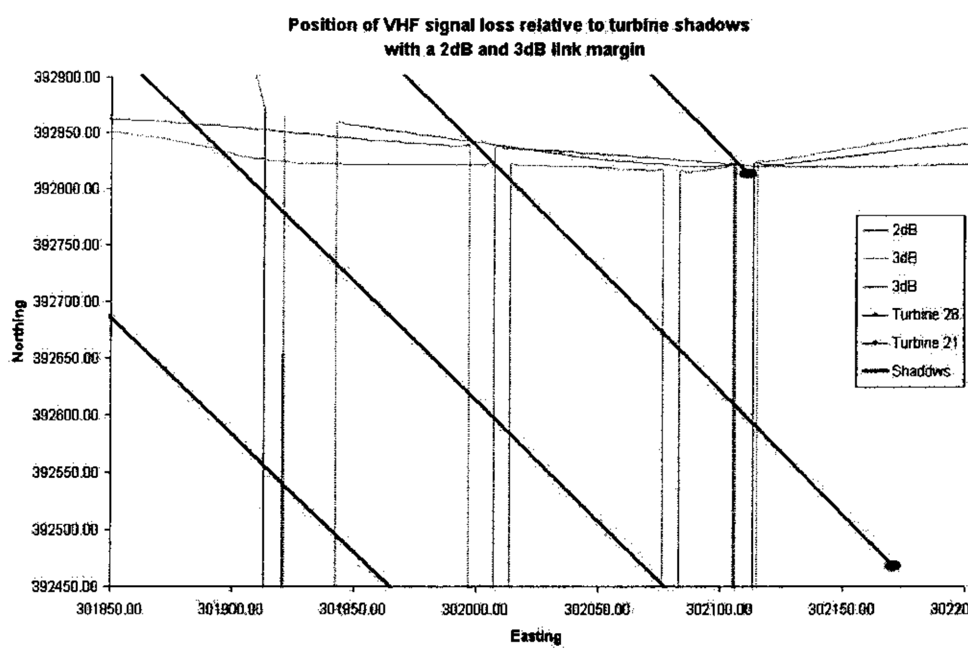



*Figure 3-2: Position of VHF signal loss relative to turbine positions with a 2dB and 3dB link margin*

Figure 3-3 shows the position at which a signal loss was observed when the link margins were 4dB and 5dB. Here the signal loss only occurs in the shadow of turbine 26. This is expected since the 2dB and 3dB link margin results (Figure 3-2) showed the shadow of turbine 21 at 500m to be only 2dB to 3dB.

A further experiment to find the depth of shadow immediately behind a wind turbine was undertaken. This test involved adjusting the link margin when immediately behind a turbine in the shadow until the signal was regained. We found that the depth of shadow at this position behind a turbine was around 10dB.

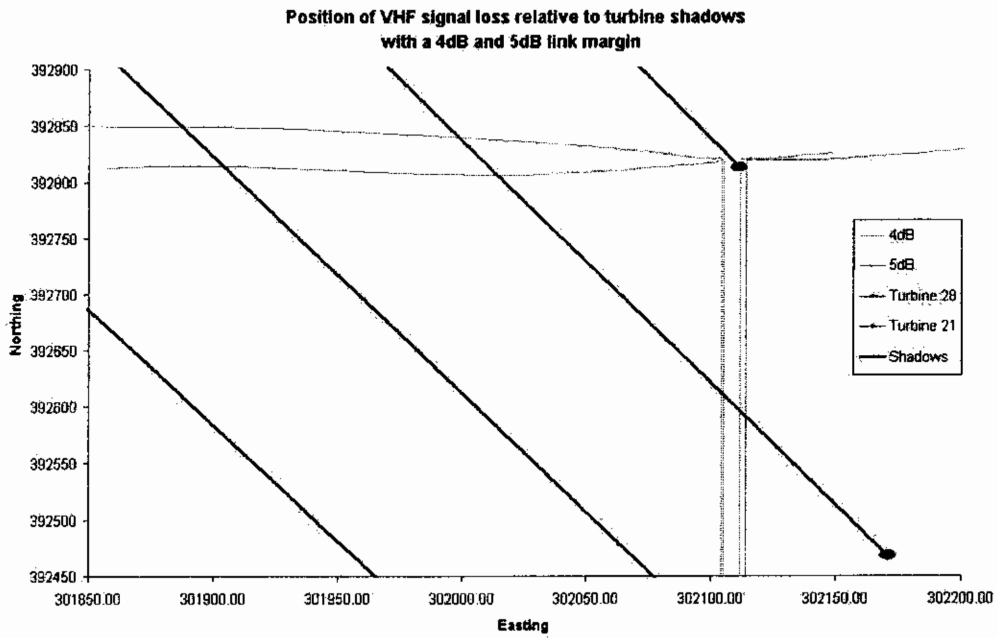



*Figure 3-3: Position of VHF signal loss relative to turbine positions with a 4dB and 5dB link margin*

## 3.3 Summary

The shadows found experimentally agree with the theoretical results outlined in the original study [1]. The affects are small and will not effect the VHF systems used in the wind farm unless the link margin between the transmitter and receiver is very low. This will only occur at long range and other effects caused by other users on the VHF channel are likely to present a greater problem.

# 4 MCA VHF communications trial

## 4.1 Overview

To evaluate the operational use of typical small vessel VHF transceivers when operated close to wind farm structures.

### 4.1.1 Equipment used

The following was required for the trial:

- A person with a hand-held VHF radio landed on a turbine platform and a vessel fitted with a typical small craft VHF radio;
- Co-operation of RNLI lifeboats, with RNLI shore stations, HM Coastguard and Mersey Docks and Harbour Board.

### 4.1.2 Method

In calm weather conditions, a person was landed on the platform of turbine 28 from the Hoylake lifeboat "Lady of Hilbre" which then moved away from the turbine. The Rhyl lifeboat "Lill Cunningham" was stationed as close to the south of turbine 3 as was safe and practical. The person on the platform positioned himself on the northerly side of the turbine tower, i.e. at the point at which the full diameter of the tower lay between him and the direction of the lifeboat.

Using VHF channel 10 and others designated for this purpose by HM Coastguard, the person on the platform transmitted in a normal conversation voice. The quality of the reception was noted by the lifeboat crew and the designated shore stations.

The lifeboat's VHF radio direction finding equipment then used this signal to determine its bearing and a comparison made with the true known bearing, any difference being recorded.

The Rhyl lifeboat then proceeded in an easterly direction on a course passing as close as was safe and practical to the other turbines on the southern boundary of the wind farm. The quality of the reception being recorded. When past turbine 1, the course was reversed, and the effects similarly noted until turbine 5 was reached. This schematic is illustrated in Figure 4-1.

The vessel's GPS positions were recorded during the whole exercise so that if any degradation of communication or direction finding is found to exist, the arcs over which this occurred could be calculated.

A principle of these tests was that, if small vessel ship-to-ship and ship-to-shore communications were not affected significantly by the presence of wind turbines, then it is reasonable to assume that larger vessels, with higher powered and more efficient systems would also be unaffected.

During this time a number of mobile telephone calls were made from ashore, within the wind farm, and on its seawards side. No effects were recorded using any system provider.

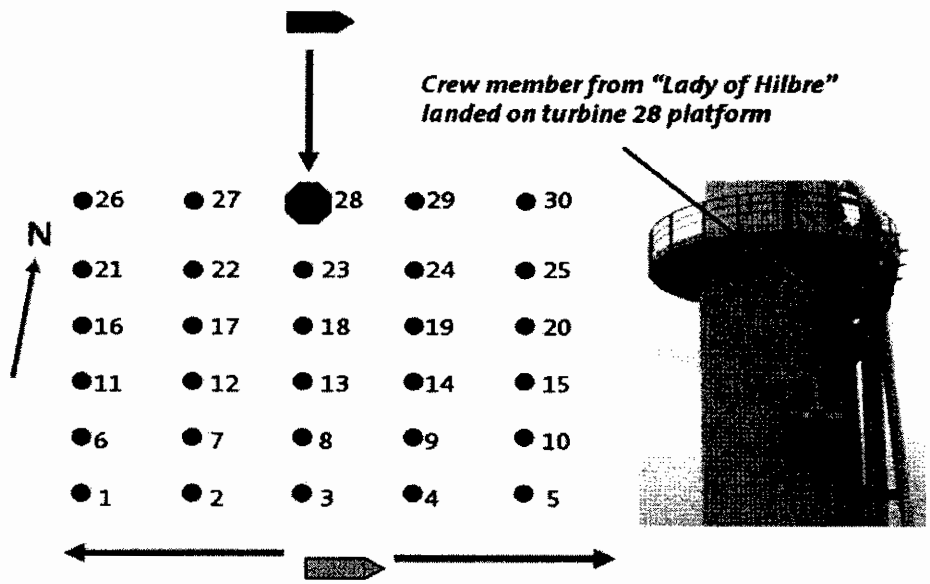



*Figure 4-1: MCA VHF communications evaluation schematic*

## 4.2 Results

### 4.2.1 VHF Communications

The wind farm structures had no noticeable effect on voice communications within the wind farm or ashore.

However, the use of the lifeboat's automatic digital direction finding equipment was severely impaired when very close to a turbine tower on the far side of which lay the transmitting vessel's direction. This was resolved when the lifeboat moved further than 50 metres from the tower.

If this effect is recognised, it should not be a problem in practical search and rescue (SAR).

### 4.2.2 Other communication methods

- **Mobile telephone communications** : There was no noticeable effect on mobile telephone communications systems.
- **Digital Selective Calling (DSC)** : The DSC system communications within the wind farm, contact being made via Holyhead and Liverpool Maritime Rescue Sub-Centres.
- **Automatic Identification System (AIS)** : AIS operated satisfactorily between vessels and as monitored by HM Coastguard MRSC Liverpool, indicated that both VHF and GPS components operated satisfactorily.

  Since it had already been determined that GPS and VHF were not significantly affected by the wind farm structures, the "Norbay" was simply asked to use her AIS when around and in the wind farm, and Liverpool MRSCC to log the reception from the ship. "Norbay" reported that she picked up other vessels' AIS transmissions without problems and Liverpool that they had similarly picked up the ship itself.

It could be argued that there might have been a ship in the area which did not receive "Norbay"s signals, or was not picked up herself by "Norbay". In view of the other evidence, however, this seems very unlikely. As noted in the Executive Summary with respect to on-going data collection, AIS-fitted vessels and HM Coastguard will report any possible omissions.

- **Telemetry Links** : The UHF telemetry link between the service vessel "Clwyd", its RIB and the BHP Billiton shore station at Gwaenysgor was reportedly interrupted when the RIB was close to turbine towers. Telemetry is normally used on fixed installations for communicating measurements such as wave and tidal heights, wind speeds, etc. However, the Radio Agency has specific requirements for short range devices that do not require licensing and may be used on marine mobiles. Any reported effects should be investigated further.

# 5 QinetiQ radar trials

## 5.1 Overview

There were two parts to the radar trials. The first dealt with the clutter effects on ship-borne radar and the second considered shadowing from wind turbines.

The radar shadow trial involved a launch travelling along a predefined course whilst being monitored by an on-shore radar at Prestatyn. The radar clutter (spurious echoes) trial used the launch "Fast Cat" to see what effect the wind turbines have on the radar display at different ranges and gain settings. Full technical details of these tests can be found in the trials plan [2].

## 5.2 Radar clutter trial results

Four different positions from the centre of the wind farm were used for the spurious radar echo trial. The first position is at the centre of the wind farm. The second and third positions are 1000m and 3000m from the centre of the wind farm respectively. The fourth position is approximately 6000m from the wind farm centre. The radar screens at each of these ranges, when using different gain settings, are shown in Figure 5-1 to Figure 5-6. In all the figures the position of the launch is in the centre of the radar display, at the bottom of the vertical line.

At the centre of the wind farm, the radar display when the gain is automatically set and manually adjusted is shown in Figure 5-1. It can be noted that the automatic gain setting is inappropriate in this case. The figure shows significant numbers of false plots (spurious echoes) of turbines and the beginning of ring-around (side lobe echoes). Using manual adjustment to reduce the gain from 60% to just 20%, the spurious echoes are almost removed entirely.

In Figure 5-2 the radar display at the second position, 1000m from the wind farm centre is presented. Here it can be observed that at a range setting of 1/2 nm there is effectively no clutter visible. However, with a 3nm range setting there is significant clutter on the radar display. In both cases the radar gain was on the automatic setting. The radar displays at position 2 illustrate how altering settings on the radar system can improve the visible output. In this case moving to a shorter range has lowered the gain. A different pulse length is also used on this range scale.

The radar displays observed at position 3 are presented in Figure 5-3. These figures show that the wind turbines are much clearer at the lower gain setting. Furthermore, in both cases there are very few false plots or evidence of side lobe break through originating from the turbines.

In Figure 5-4, Figure 5-5 and Figure 5-6 the radar screens observed with gain settings of 64% (automatic setting), 54%, 44%, 34% and 24% are shown. It can be noted that the turbines are visible as discrete plots. The large region of clutter is the coastline. As the gain is reduced, the wind turbines remain on the screen although by a gain of 34% a number of the turbines have disappeared. With a gain setting of 24% the number of visible turbines has reduced significantly. It is interesting to note that the turbines that do disappear are turbines that are shadowed by other turbines. A further consequence of reducing the gain is that small targets at long range may no longer be detectable.

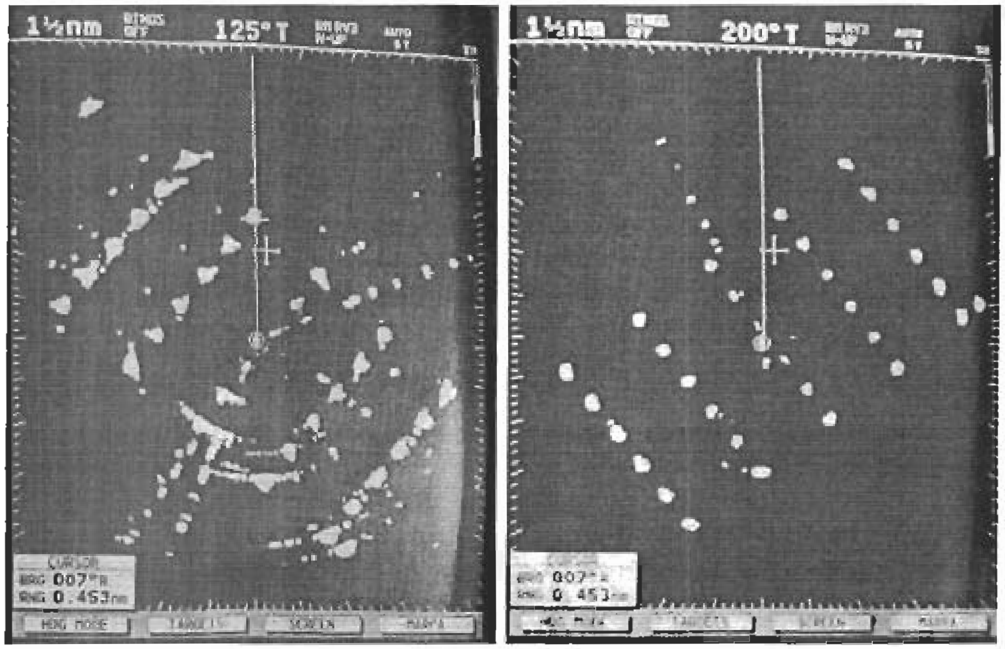



*Figure 5-1: Position 1, the wind farm centre, with gain settings of 60% (left) and 20% (right)*

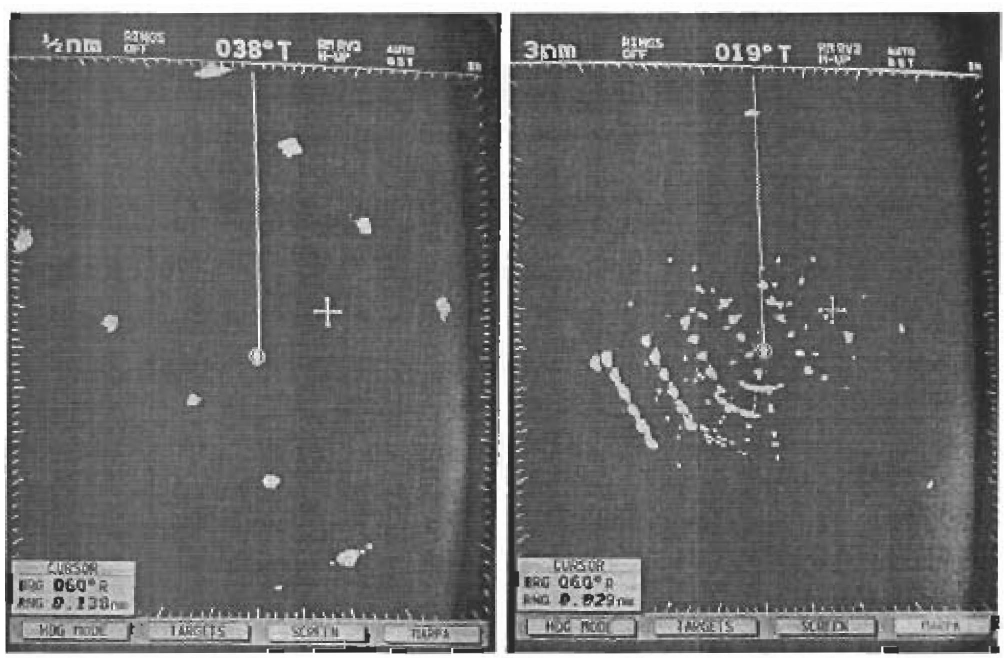



*Figure 5-2: Position 2, 1000m from wind farm centre, close up (left) and the whole wind farm (right) with an automatic gain setting*

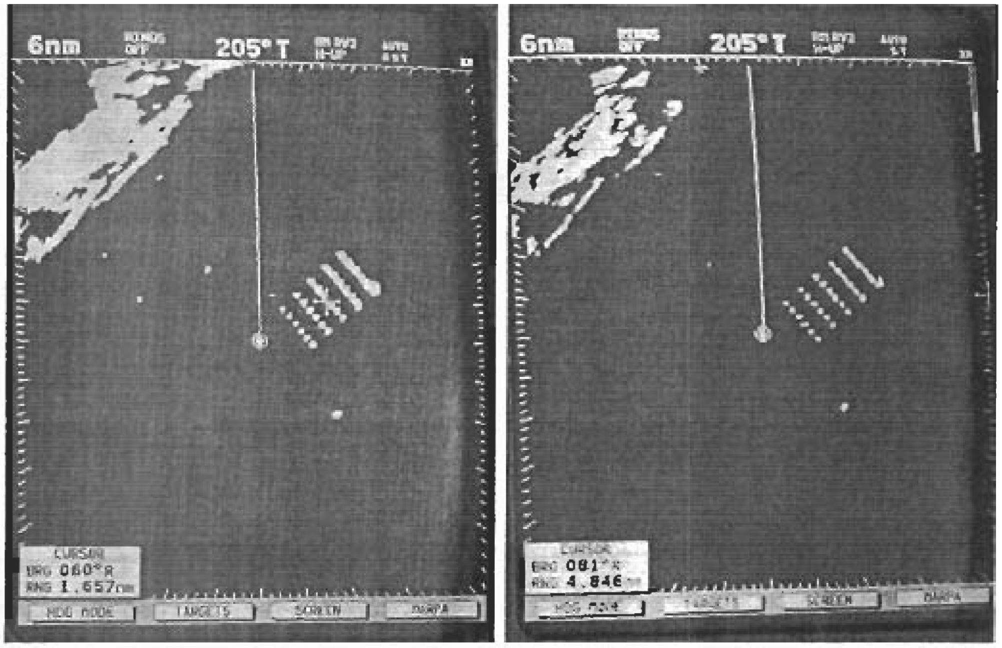



*Figure 5-3: Position 3, 3000m from wind farm centre, 74% gain (left) 44% gain setting (right)*

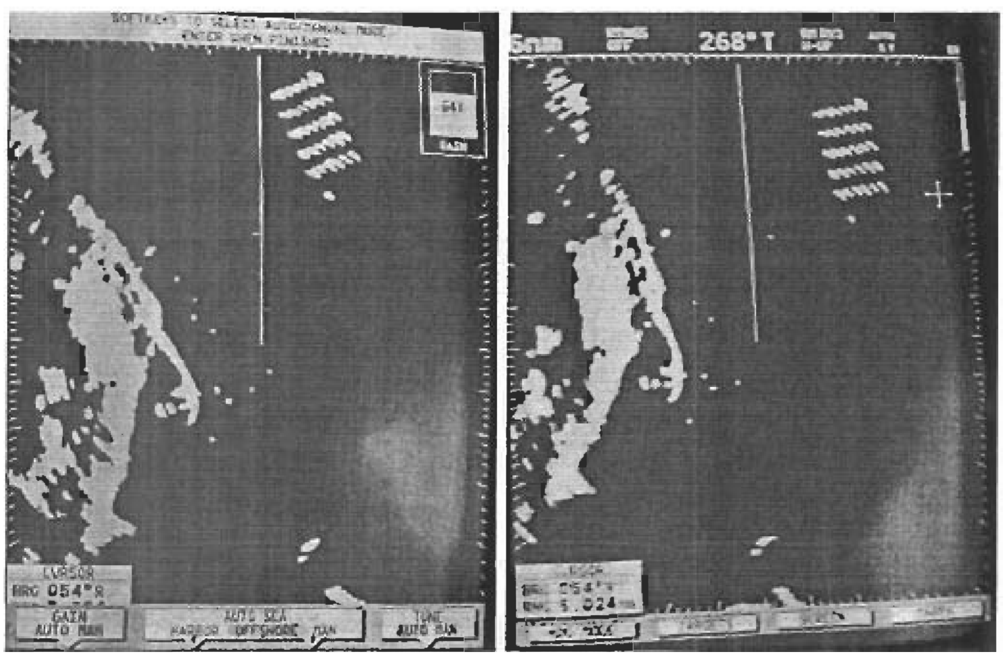



*Figure 5-4: On route to the wind farm with 64%(left) and 54%(right) gain settings*

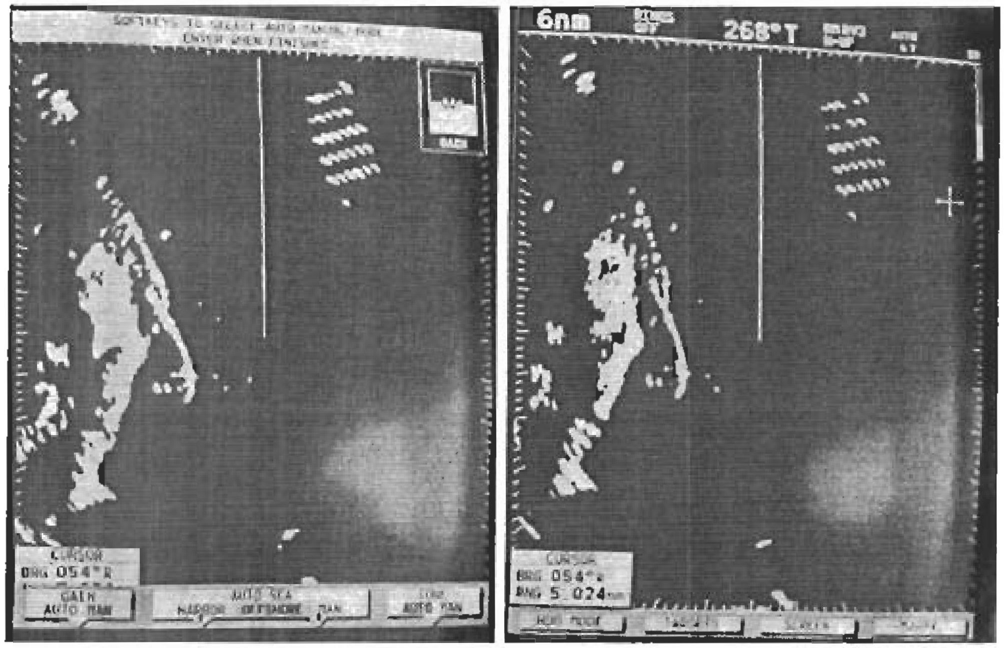



*Figure 5-5: On route to the wind farm with 44%(left) and 34% gain settings*

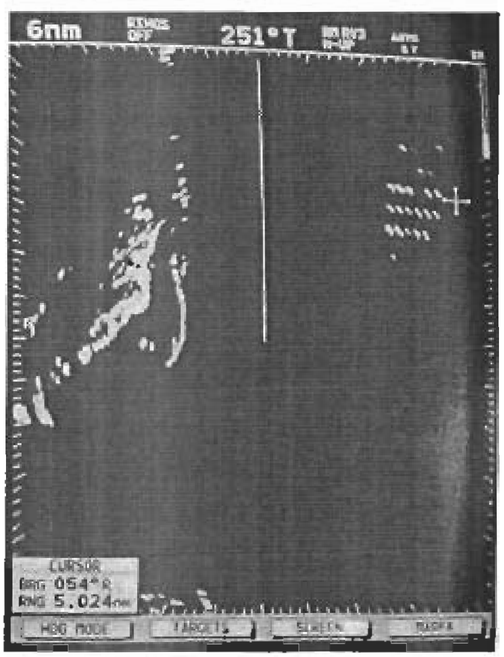



*Figure 5-6: On route to the wind farm with 24% gain setting*

## 5.3 Radar shadow trial results

As outlined above and in more detail in the trial plan, the radar shadow trials involved monitoring the radar display of a shore based radar at Prestatyn. Specifically, the purpose of the trial was to look for signal loss of the target boat, due to the presence of wind turbines. Shadows at the radar frequency of 9.4GHz are deeper than those seen at VHF frequencies (150MHz).

If we consider the gain settings of the radar then an estimate of the shadow depth can be gauged.

The peak power of the radar is 4kW which corresponds to 36 dB. Assuming a log adjustment to the gain we find that, for example, at 54% gain the power is 19.44 dB. With a gain setting of 54% or 19.44 dB the wind turbines were visible. However, reducing the gain to 44% or 15.84 dB we found that the unshadowed turbines were still visible, but the shadowed turbines had disappeared from the display. The distance behind the shadowing turbine was approximately 1000m. A further reduction of the radar gain to 4% or 1.44 dB, it was found that the unshadowed turbines began to disappear. This can be seen in Figure 5-4 to Figure 5-6

From these observations we find that the difference in power required to detect an shadowed (1000m behind a shadowing turbine) and unshadowed turbine is approximately 14.4 dB.

At 1000m the theoretical study[1] suggests that the shadow depth behind a wind turbine is approximately 14.5 dB, which agrees very well with the estimate made using the radar displays and radar gain settings.

## 5.4 Summary

There were two parts to the radar trials. The first dealt with the clutter effects on ship-borne radar and the second considered shadowing from wind turbines.

In the first trial it was found that adjusting the radar gain could reduce the number spurious echoes significantly. However, a consequence of gain reduction is that small targets at long range may no longer be detectable. And at very low gain settings (approximately 34% or less) some shadowed wind turbines start to disappear.

The second part to the trial dealt with radar shadows behind wind turbines. It was found that the depth of shadow at a distance of 1000m behind a turbine was approximately 14.4dB. This value was consistent with those determined in theoretical studies undertaken previously [1].


MCA and QinetiQ proprietary

# 6 MCA Radar trials

## 6.1 Overview

The wind turbine generators (WTG) are very large structures in the vertical plane and significantly so in the horizontal plane. Although the towers are cylindrical, their diameter of 5 metres and height above the water - around 70 metres - is such that they have a comparatively large reflecting surface area. This is compounded by the reflecting surfaces of the platforms, ladders and other structural features of the towers, an average total of about 80 square metres of signal returning surface at any time and from any direction. The three bladed rotors have a total reflecting area of around 200 square metres when their plane is at right angles to the direction of the radar scanner, and around half that when in line with it. The nacelle and boss have reflecting areas of up to 16 square metres. Thus in the vertical plane the North Hoyle WTGs can have a radar signal returning area of around 300 square metres. The sections of turbine which are other than at right angles to the shipborne radar, i.e. non-returning, may produce reflected and other spurious echoes. The scale of the structures is better illustrated in Figure 6-1.

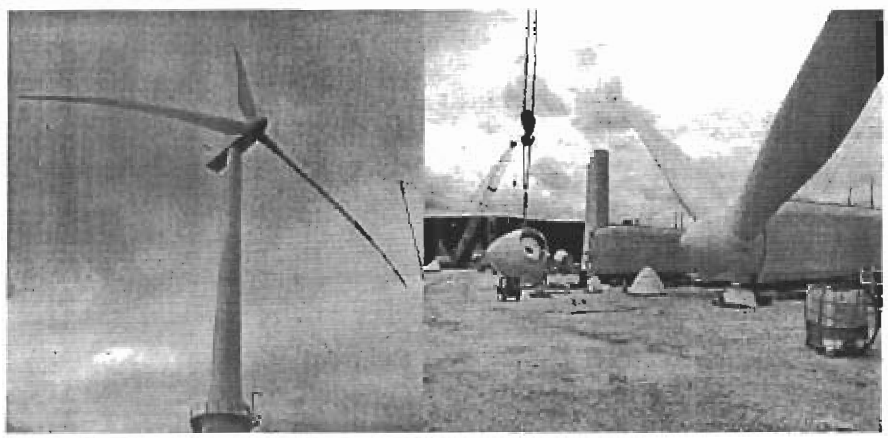


*Figure 6-1: North Hoyle Vestas wind turbines*

This is a critically important factor when shipborne or VTS radars are close to the WTGs. Here the vertical beam width, for most ships' radars this being between 25 and 30 degrees, has a greater effect than the horizontal beam width, usually between 1 and 2 degrees.

When close to turbines, the response from individual transmitted pulses may therefore be significantly greater than if, for example, at the same range from a large ship which would be unlikely to have an equivalent vertical extent.

This has some advantages in, for example, detecting wind farm structures by radar, but can have disadvantages with respect to the use of radar in SAR, automatic radar plotting aids (ARPA), collision avoidance or vessel traffic services (VTS). It will also have implications for the siting of radar beacons (RACONS).

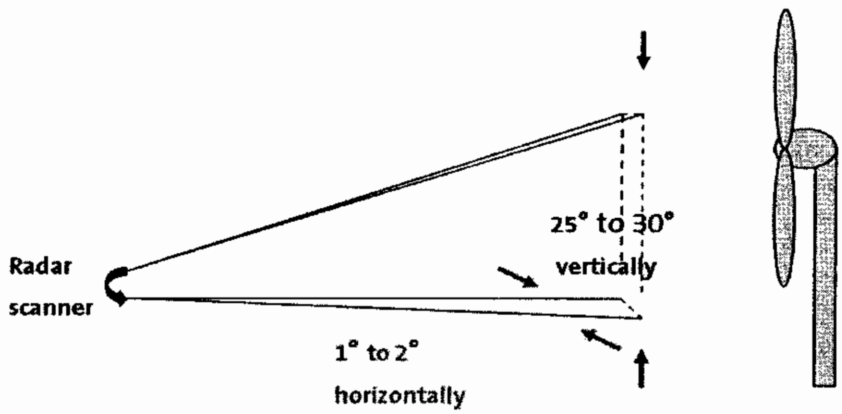



*Figure 6-2: Typical radar scanner horizontal and vertical beamwidths*

As the radar station increases in distance from the wind farm, this effect reduces in significance. For example, as will be seen in subsection 6.16.1, at the range of the Mersey Docks and Harbour Board's Seaforth radar from the wind farm, 14 nautical miles (nm), the vertical extent of the WTGs has little effect and larger vessels such as the "Norbay" (17,464 Gross Tons ) could be detected and tracked. Smaller vessels, such as the lifeboats and service craft could not be detected at this range.

Technical details of all the radar systems used by the MCA during the trials can be found in Appendix B.

This report is not intended to explain marine radar systems or their operation. A number of publications are available that deal with this and other marine navigation subjects. An example is suggested in reference [4].

## 6.2 Small vessel radar evaluation

### 6.2.1 Overview and method

To evaluate the operational use of typical small vessel radar systems when used to detect vessels within and close to wind farms.

With the Rhyl lifeboat "Lill Cunningham" lifeboat stationary very close to the northern side of turbine 3, the Hoylake lifeboat "Lady of Hilbre" traversed the wind farm on a track midway between the turbine rows 10 to 6 and 15 to 11, on a straight line course parallel to these towers. The vessel then proceeded to the south of turbine 21 and similarly passed between the rows 16 to 20 and 21 to 25. Finally, the vessel proceeded to a point 250m north of turbine 30 and followed a course parallel to the northern boundary of the wind farm. The stationary "Lill Cunningham" at turbine 3, fitted with the video camera, with the radar set on the 3 nautical miles range, recorded the displayed data. The data was analysed to determine the blind arcs and shadow areas produced by turbine 3 and others in the wind farm. The courses followed are illustrated in Figure 6-3 and pictures of the life boats used are shown in Figure 6-4 and Figure 6-5.

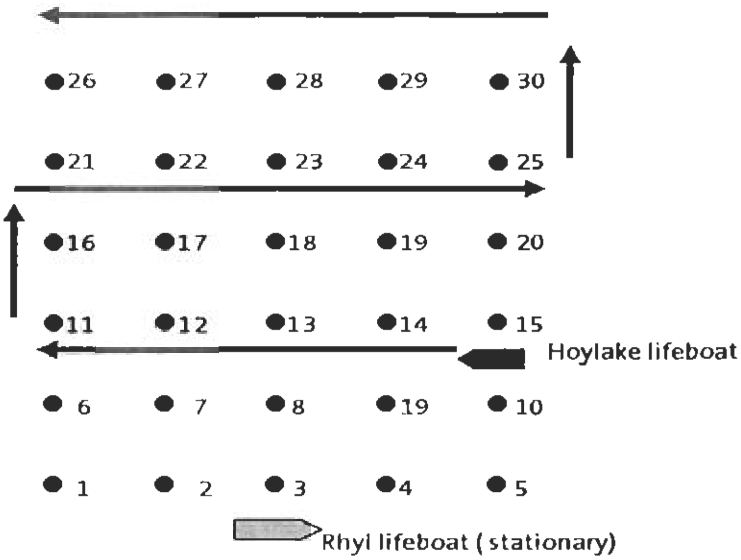



*Figure 6-3: MCA small vessel radar detection capabilities schematic*

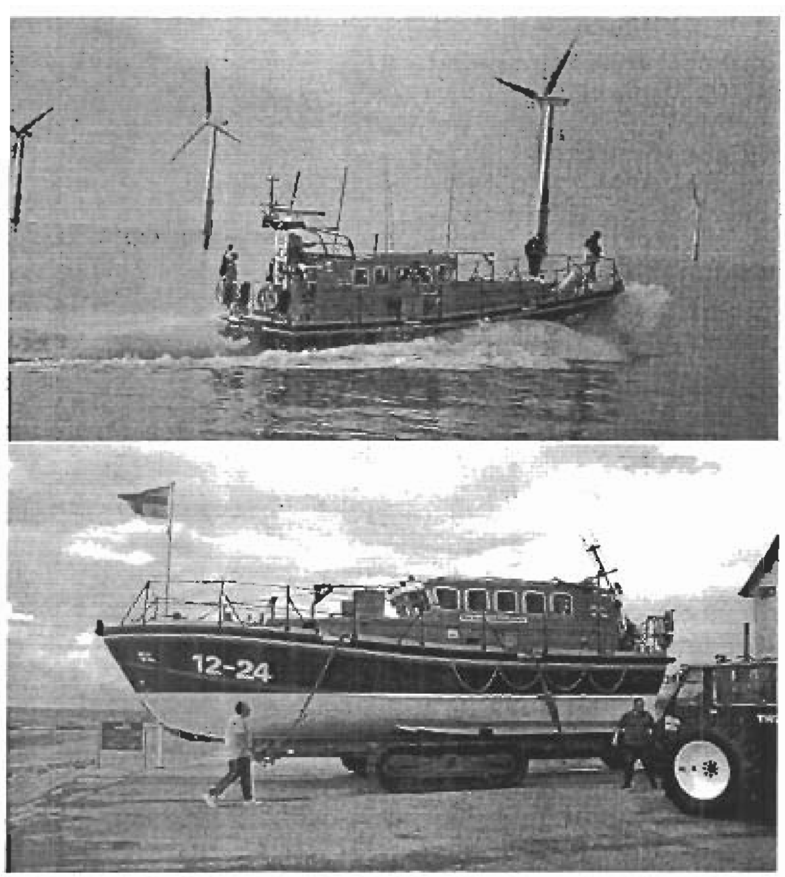



*Figure 6-4: The Hoylake RNLI lifeboat "Lady of Hilbre" (top) and the Rhyl RNLI lifeboat "Lill Cunningham" (bottom)*


*Figure 6-5: The Rhyl RNLI inshore lifeboat*

## 6.3 Results of the trials

### 6.3.1 Shadow and blind areas

As has been noted previously, the WTGs produced blind and shadow sectors behind them in which other turbines and vessels could not be detected and displayed. An example of this is illustrated in Figure 6-6. Additionally, the strong response of the WTGs when nearby, and with their close spacing, appears to produce saturation areas in which targets are not detected, particularly if receiver gain is reduced to reduce side lobe and other spurious echoes. However, in general, this would only be a significant problem if:

- the search vessel or target were not able to move to different locations from where the target was not in these sectors;
- the target lay within the poor cross and down range discrimination areas of the WTG responses, as illustrated in the following trials.

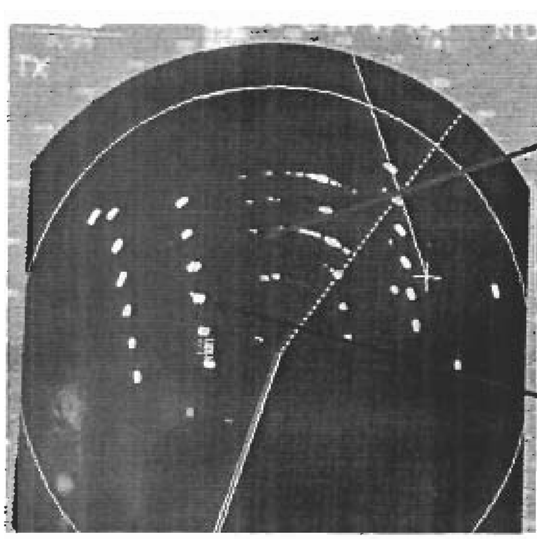
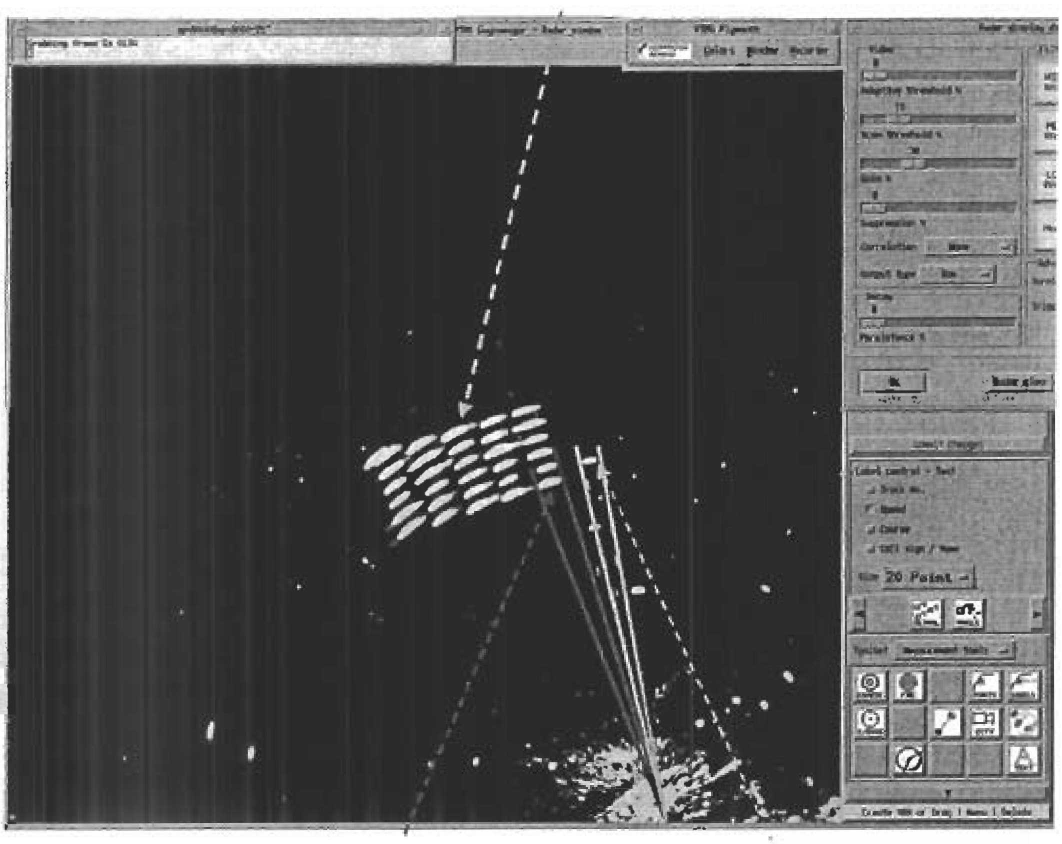



*Figure 6-6: Shadow and blind arcs with side lobe echoes*

## 6.4 Principles of range and bearing discrimination testing

The effect of turbine blades on turbine echo size is illustrated in Figure 6-7, where the plane of the rotor blades is approximately at right angles to the direction of the radar scanner. Here the angular width of the turbine is 1.6 times that of the anemometer mast. Corresponding sizes of the echoes displayed at the relevant ranges are about 610 metres and 300 metres respectively. The displayed size of turbine and anemometer mast is $2\tan(\theta/2) \times R$, where $R$ is the range in metres and $\theta$ is the angle subtended by the displayed echo. The displayed range discrimination is approximately 200 metres.

***Range discrimination determined by turbine down-range echo depth measured at specified pulse lengths***




*(Angles not to scale)* ***Bearing discrimination of targets close to turbine is determined by theta (angular width of turbine) at a given range.*** ***Anemometer mast***

*Figure 6-7: Range and bearing discrimination*

## 6.5 Range discrimination test one

### 6.5.1 Method

With Hoylake lifeboat "Lady of Hilbre" stationary, alongside turbine 1, on its Northerly side, Rhyl lifeboat "Lill Cunningham" maintained a Northerly course towards turbine 1. With the radar initially set on its 6 nautical miles range and using a video recorder, the display was recorded continuously from a distance of 4 nautical miles from turbine 1. Additionally, it was noted whether and at what range, if any, the echo of target vessel "Lady of Hilbre" could be visually resolved from the return from the turbine. As "Lill Cunningham" approached turbine 1 the radar was progressively set to shorter ranges and pulse lengths.

It should be noted that the initial four nautical miles range was chosen since it was a fair representation of the range at which search and rescue activities would be fully under way. The track followed is in Figure 6-8.

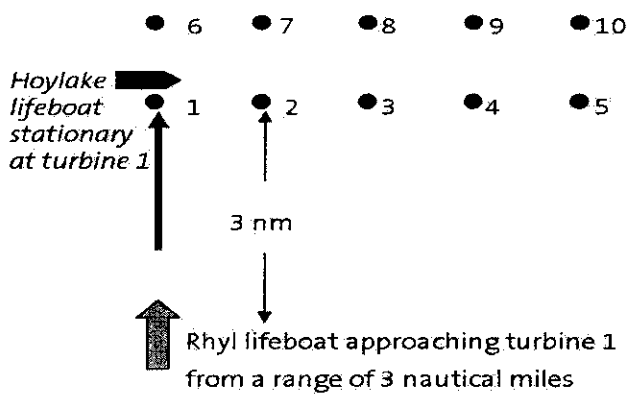



*Figure 6-8: MCA range discrimination test 1 schematic*

### 6.5.2 Results of the trial

As the "Lill Cunningham" approached the wind farm, the echo of "Lady of Hilbre" could not be seen to separate from that of turbine 1. This is shown in Figure 6-9. With "Lill Cunningham" 1.5 nm from turbine 1 and "Lady of Hilbre", 30 metres west of turbine 1 and 25 metres down range from it, the radar was put on a 3nm range, short pulse setting. It can be seen (see Figure 6-10) that there is no echo separation. The anemometer mast, approximately 170 metres to the west of turbine 26, is not separated in azimuth from it due to beam width effects.

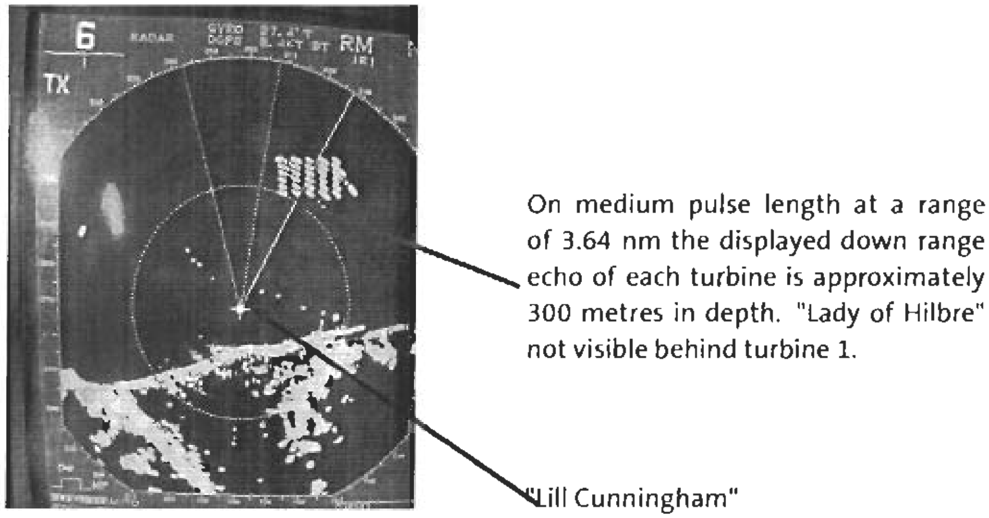



*Figure 6-9: "Lady of Hilbre" in turbine shadow on 6 nm range*

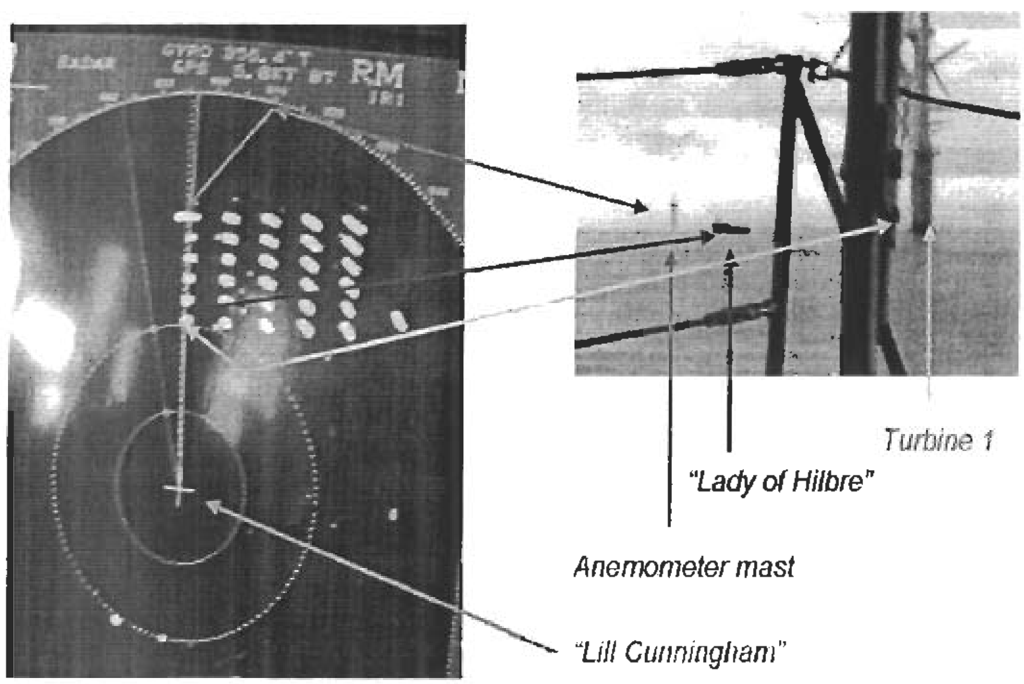



*Figure 6-10: Still in shadow on 3nm range*

## 6.6 Range discrimination test two

### 6.6.1 Method

Since there was no down-range separation of the echo of "Lady of Hilbre" from that of the turbine on these radar ranges, then the following trial was carried out with "Lill Cunningham" initially stationary 3 nautical miles to the south of turbine 1, its radar set to the 3 nautical mile range and "Lady of Hilbre" very close to turbine 1. "Lady of Hilbre" headed slowly towards turbine 6, the object being to note where its echo clearly separated from that of turbine 1 on "Lill Cunningham"'s radar . This separation was however not observed. Therefore, a series of runs were performed by "Lady of Hilbre" while "Lill Cunningham" slowly proceeded towards turbine 1. The courses followed are illustrated in Figure 6-11.

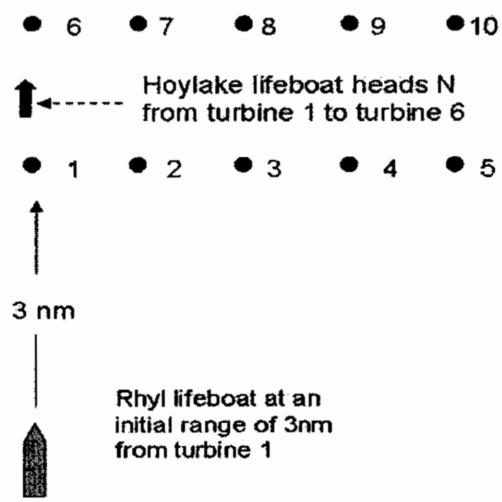



*Figure 6-11: MCA range discrimination test 2 schematic*

### 6.6.2 Results of the trial

While "Lady of Hilbre" remained in the shadow of turbine 1, no echo was received. However, when she kept on a line 30 metres to the west of that joining turbines 1 and 6, the echoes separated at a down range distance of some 200 metres from turbine 1, when "Lill Cunningham" was 1.4 miles from turbine 1, radar set to 1.5 miles range, short pulse, and with the gain control turned down to reduce side lobe and reflected echoes. The observed range discrimination is shown in Figure 6-12.

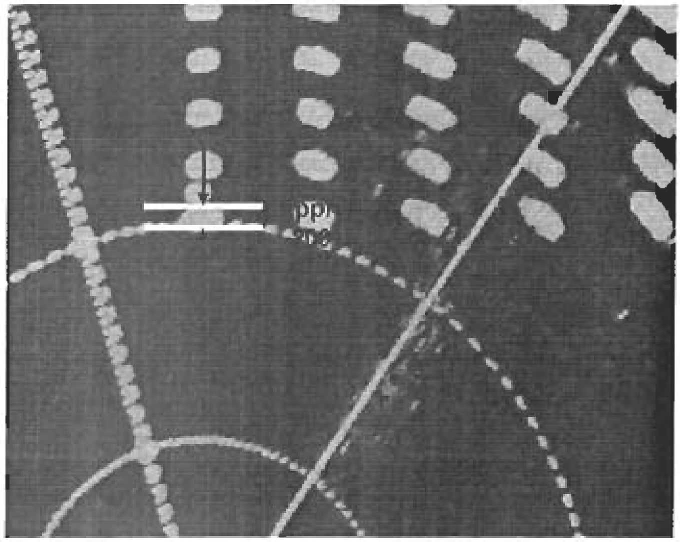


*Figure 6-12: Observed range discrimination*

## 6.7 Bearing Discrimination

### 6.7.1 Objectives and method

The objectives of these trials are similar to those of range discrimination, but in azimuth rather than down range.

Hoylake lifeboat traversing East and West of turbine 1, with "Lill Cunningham" stationary 3 nautical miles South of turbine 1, its radar set to the 3 nautical mile range and "Lady of Hilbre" very close to the northerly side of turbine 1, the size of the cross-range arc of the returned echo of turbine 1 was measured using the radar's bearing markers. The course is illustrated in Figure 6-13.

"Lady of Hilbre" could not be visually distinguished from the echo of the turbine therefore proceeded slowly on a westerly course until its echo on "Lill Cunningham"s radar visually separated from that of the turbine. "Lady of Hilbre" then proceeded on a reciprocal easterly course until its echo on the radar on "Lill Cunningham" again separated from that of the turbine. Radar bearings and ranges of "Lady of Hilbre" were recorded at both of these instances. The full procedure was recorded by video camera. It should be noted that the radar beam width, unlike pulse length, will not vary significantly with the range to which the system is set and thus, the bearing discrimination in degrees will be effectively a constant. Cross-range response widths can be calculated for other ranges from the turbines at which the search vessel ("own ship") may lie (see Figure 6-7).

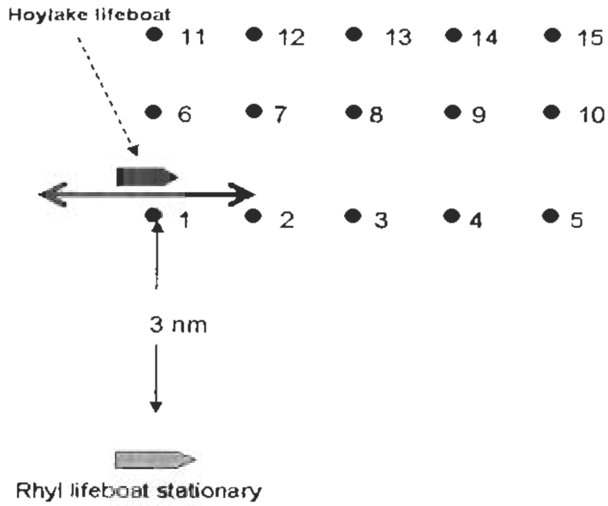



*Figure 6-13: MCA bearing discrimination test schematic*

### 6.7.2 Results of the trial

Full separation both west and east of turbine 1 was achieved at an angle of 4 degrees at the observation range of 3 nm. This angle is measured from the centre of the turbine echo to the centre of the target echo and equates to a distance of 388 metres.

It should be noted that the target would only show as a distinct and separate echo when some 385 metres clear of the turbine tower and therefore it would not be detectable for a distance of 770 metres from one side of the turbine to the other. As can be appreciated, the echo of a target travelling through this turbine array would be separate from nearby turbines and trackable by ARPA for only short periods of time and distance.

The results are illustrated in 6-14.

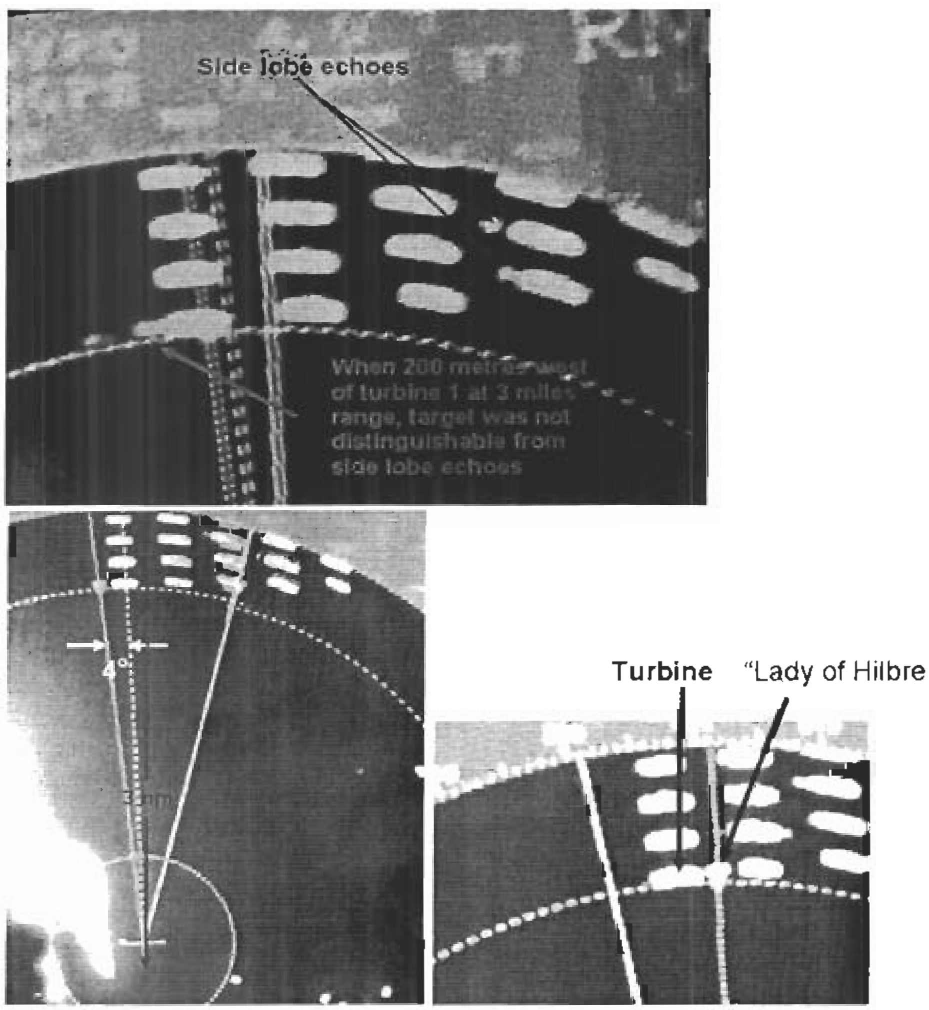



*Figure 6-14: Bearing discrimination trials results*

## 6.8 Down and across range target discrimination

### 6.8.1 Overview

The problem here relates to the scanner beam width and pulse length in use. Theoretically the across range size ( in metres) of a displayed target is equal to its beam width at that particular range from the target plus twice the cross range target size, i.e:

$$W = 2\tan(\theta/2) \times R_{target} + R_{cross}, \quad 6\text{-}1$$

where $W$ is the beam width in metres, $\theta$ is the horizontal beam width angle, and $R_{target}$ and $R_{cross}$ are the target range and target cross range sizes respectively.

Echo depth in metres is equal to half the pulse length in microsecs, times the speed of propagation of radio waves, plus the down range depth of the target, which can be expressed as:

$$D_{echo} = (p \times 300/1\mu s)/2 + D_{target}, \quad 6\text{-}2$$

where $D_{echo}$ is the echo depth in metres, $p$ is the pulse length in $\mu$s and $D_{target}$ is the target depth.

However, the displayed sizes of the North Hoyle WTGs from Gwaenysgor are significantly greater than that, the across range echo size being around 600 metres at a range of 5.2 nm and the down range depth being around 200 metres.

The across range effect is due to the fact that, since the vertical extent of the turbines is large, when the transmitting vessel is close they will return power outside the nominal beamwidth of the radar. That is, the response will include significant power from outside the half power (-3dB) points of the main beam.

This has two effects, firstly that a vessel initially close to the turbine will not be detected until it has moved some hundreds of metres across range or a smaller distance down range. Additionally, the effects of side lobes. shadow and blind sectors and multiple or reflected echoes may compound these ranges.

For ARPA or VTS / Port radar tracking systems the effects are likely to be that tracking vessels within or close to wind farms is difficult. This was found to be the case with the "Norbay" ARPA systems and with the BHP Billiton tracking system at Gwaenysgor.

## 6.9 Side lobe, reflected and multiple echoes

The objectives of this part of the trials were to examine the potential effects of spurious echoes on target detection and general navigation in the vicinity of the wind farm.

With Rhyl lifeboat "Lill Cunningham" 50 metres WSW of turbine 1, Hoylake lifeboat "Lady of Hilbre" proceeded on a straight line course parallel to the boundary line of turbines 1 to 5 and 50 metres from each turbine, commencing at turbine 5 (as shown in Figure 6-15). "Lill Cunningham" used her radar set to the shortest relevant ranges with normal gain settings and any side lobe, multiple or reflected echo effects were recorded. The results can be seen in Figure 6-16.

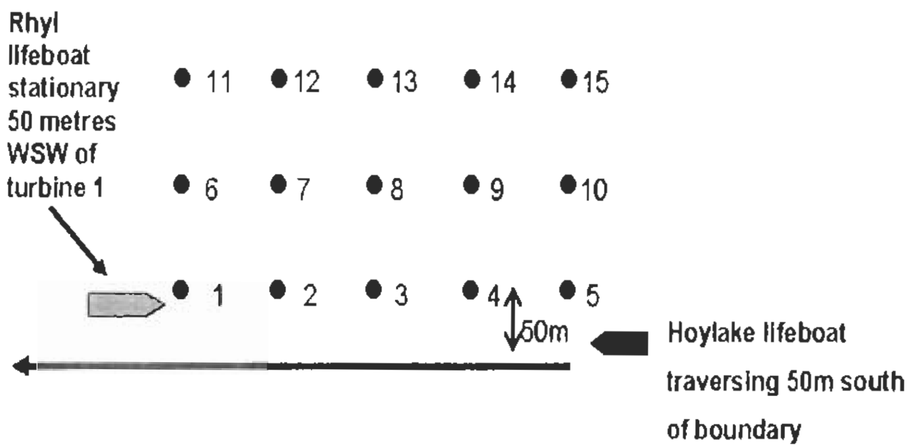
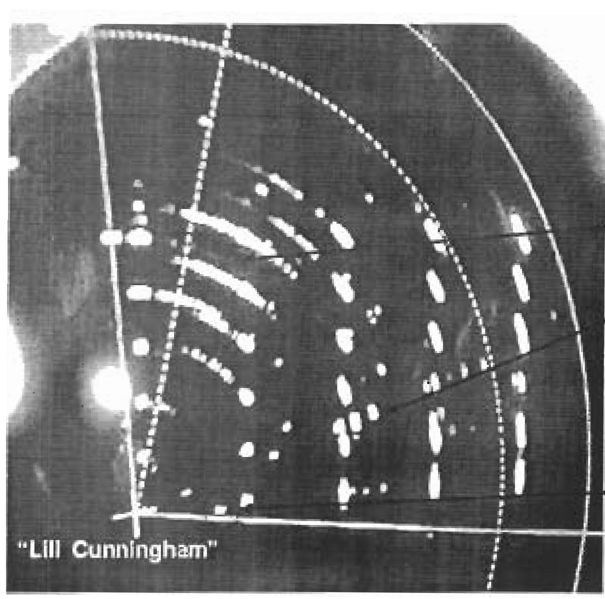



*Figure 6-15: MCA Schematic for assessing side lobe, multiple and reflected echo effects*




As with all ranges closer than about 1.5 nm, side lobe echoes and multiple echoes were very strong.

Side lobe echoes

Multiple echoes

Detection of small targets or buoyage would be extremely difficult in these circumstances.

"Lady of Hilbre" at a range of 0.3 nm (550 m)

*Figure 6-16: Radar on 0.75 nm range and short pulse*

## 6.10 Further side lobe, reflected and multiple echoes identification

### 6.10.1 Objectives and method

The objectives of this trial were two fold. Firstly the spurious echoes inside the wind farm were to be examined and secondly the response of "Lill Cunningham" to a shore based radar were to be recorded (see subsection 6.12 for details of this).

"Lill Cunningham" was to proceed north between turbine columns 1 to 26 and 2 to 27. This is shown in more detail in Figure 6-17.

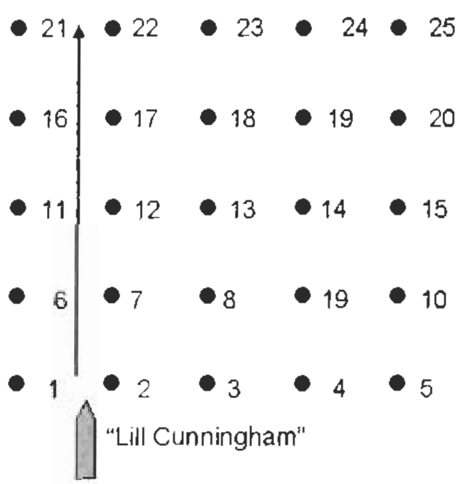



*Figure 6-17: Further side lobes schematic*

### 6.10.2 Results of the test

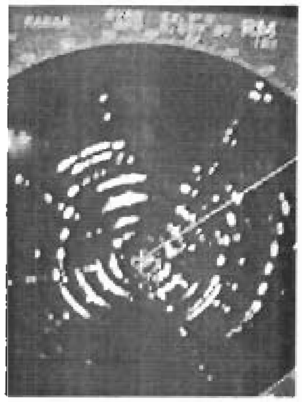
With the set tuned correctly and with proper brilliance levels, the gain control was adjusted to various levels. Within the wind farm it was found that , with the radar set on the 1.5 nm range, ie. a shorter range than the length of the wind farm site, and on short pulse, significant quantities of spurious echoes were produced at all gain levels.

i With the gain level set higher than its optimum on this range the display was severely affected by side lobe echoes.

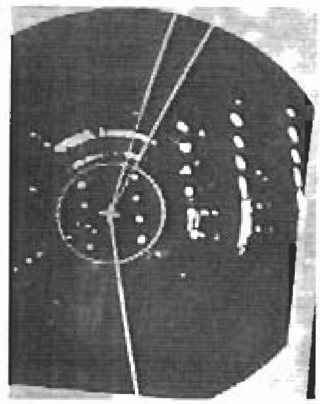
ii, iii The gain control set at its mid level, either manually or by use of the automatic gain control, would be the unit's normal level. Turning gain down to further reduce side lobe or multiple echoes would affect the detection of smaller target vessels or buoyage.

iv With gain levels approaching zero, side lobe echoes were reduced to a minimum but, with this very low level of signal amplification, small targets and buoyage would be very difficult - if not impossible - to detect.

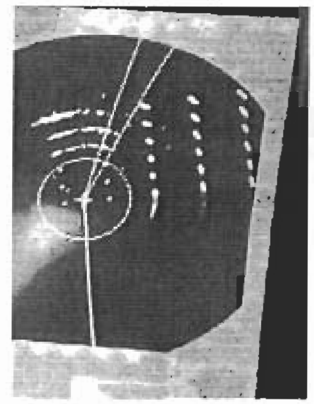
The photographs in Figure 6-18 illustrate the effects on side lobe echoes of reducing gain manually and that obtained using the automatic gain control. It should be noted that the use of swept gain anti-sea clutter controls would also reduce gain at a specific distance from the observing vessel.



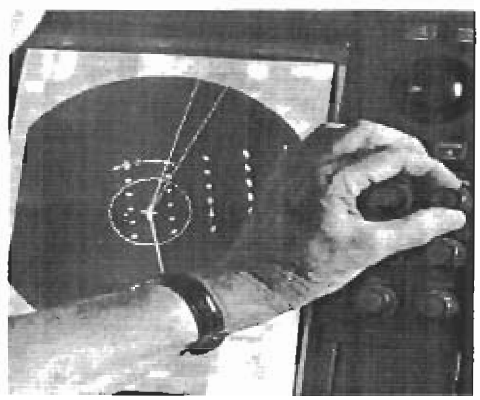
(i) Two thirds gain



(ii) Half gain



(iii) Automatic gain control



(iv) one tenth gain

*Figure 6-18: Adjusting gain levels*

### 6.10.3 Summary

Since marine radar scanners are not perfect directional propagators some emissions occur in directions other than the main beam. These are not usually critical unless strongly reflecting surfaces are in close proximity, when spurious echoes may be received from directions other than that of the main radar beam.

These were found to occur in a number of radar systems at ranges of less than 1.5 nm (2800 metres) from the wind farm. This happened in both the X band and S band type tested and approved radars carried in the "Norbay". The effects were greater on S band (See subsection 6.14).

At a range of 0.6 nm (1100 metres) from the turbines "Norbay" reported very heavy spurious echoes on S-band radar.

This effect was also examined on the X band radar of the Rhyl lifeboat "Lill Cunningham". Within the windfarm where the maximum distance from the nearest WTG is always less than 430 metres, the side lobe effect with normal gain levels was very heavy.

This would make the detection of other craft or buoyage difficult, and impossible in some conditions.

Reducing gain levels would reduce side lobe effects but would also reduce the response of those vessels for which a lifeboat might be searching, or from which other craft might be seeking to keep clear.

The experience of the "Lill Cunningham" was that, to reduce side lobe effects to zero, the gain had to be set at its minimum level. At this level small craft would not be detected, especially if they were close to WTGs (see shadow areas and bearing / range discrimination in subsections 6.2 and 6.4), in rain, or in sea clutter.

Setting the gain control at its mid level or applying the automatic gain control when less than about 500 metres from WTGs resulted in a significant proportion of spurious echoes.

For RNLI vessels' search and rescue (SAR) operations this has obvious implications. For other vessels there could be problems in collision avoidance. This would apply particularly to large or high speed vessels in which there might be a requirement to keep radars on longer range scales and with normal gain levels, when in the vicinity of wind farms, so as to plan required manoeuvres in ample time.

This would apply particularly to vessels within higher density shipping lanes which might be near to larger Round 2 offshore wind farms, and which might have joining or crossing traffic or buoyed waypoints.

MCA have proposed that a research project should be undertaken to look at improvements in the detection and discrimination of small targets, supporting the need highlighted at IMO NAV 50 in June 2004, following high-profile incidents such as the loss of the High Speed Craft (HSC) "Sleipner", in which there were sixteen deaths. It might be possible to use the results of this project to examine the overall effects of offshore wind farms on the detection of small craft, obstructions and buoyage. This could also provide further guidance to the clearance of wind farm boundaries from traffic routes or from critical buoyage and its data could be included in the proposed DTI navigational risk assessment methodology referred to in the Executive Summary

New international standards for type tested marine radars will become available after 2008. The effects of offshore wind farm structures on these will need to be assessed.

## 6.11 Sea and rain clutter within the wind farm

High winds and swell will produce sea clutter within the wind farm which will itself interfere with the detection of targets. The presence of WTGs against which waves might break may increase the overall sea clutter, which can be reduced by the swept gain control on basic radar equipment. Again, however, the reduction of gain may reduce detection and tracking abilities.

Tripod foundations may produce greater sea clutter than monopiles.

Rain clutter is produced by reflection from water droplets and, again in simple radar systems, its effect is reduced by employing fast time constants (FTC). There is generally a noticeable reduction in detection abilities when FTC is employed. An example of a radar display showing rain clutter near to the wind farm is shown in Figure 6-19.

At all times when the trials were being undertaken, there were light winds, calm seas

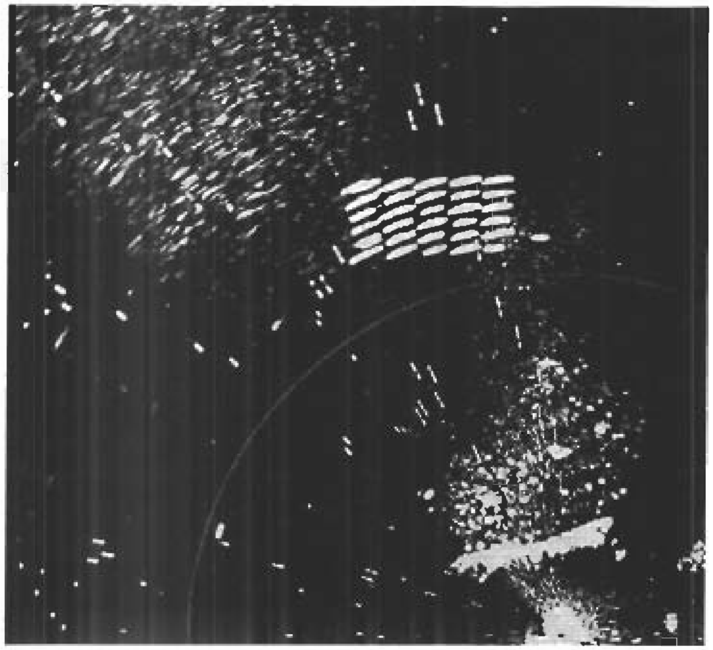


*Figure 6-19: Precipitation effects*

and clear visibility. This had some advantages in that the vessels involved were able to look at effects close to turbines. However, because of these conditions, the effects of sea clutter and precipitation in combination with the wind farm's own interference effects were not able to be examined.

Benchmarks for the range of first detection in clutter conditions are to be included in the MCA project mentioned above, clutter environments for both sea state and rainfall and as combinations of these being defined.

## 6.12 MCA tests on the effects of wind farm structures on shore based radars

### 6.12.1 Overview

The objectives were to inform the operation of VTS and Port approach radar systems in the vicinity of offshore wind farms.

Two radar systems were used in these trials, one being the mobile radar unit kindly loaned to the MCA by the Environment Agency and the other being the radar unit at Gwaenysgor, above Prestatyn. This unit is used by BHP Billiton to monitor traffic around the Douglas oil field and the Hamilton gas field, these being sited some 7.5 nm north of the North Hoyle wind farm.

Raw and filtered radar data were recorded by the Denbridge Marine APX-8000 system.

"Lill Cunningham" and "Lady of Hilbre" carried out the exercises described in the foregoing on July 21st and 22nd 2004, testing their on board systems to determine if they were degraded in any way by the wind farm. During this time, their movements were being monitored and recorded by shore based radars. The shore radar sites were

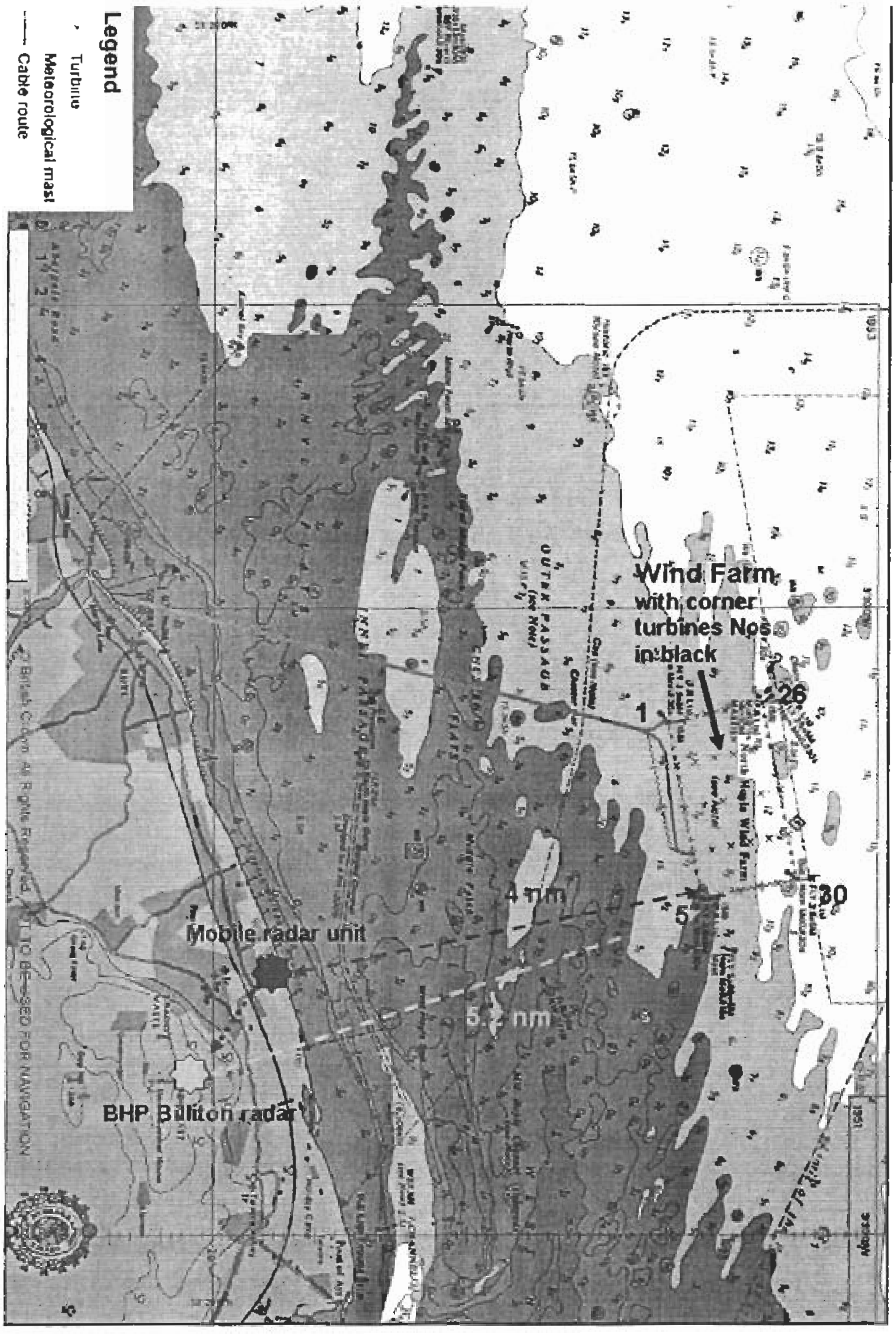



*Figure 6-20: North Hoyle wind farm with radar positions (Not to scale)*

as illustrated in Figure 6-20, the mobile radar first being located at a site almost in line with turbine column 5 to 30 and then being relocated near to the BHP Billiton radar. The recording equipment was, on the following day, then transferred from the mobile radar unit to the BHP Billiton unit.

The mobile radar was first sited along the promenade and access road next to the Prestatyn yacht club, where it had a scanner height of approximately 6 metres above sea level and was 4 nm from the wind farm.

### 6.12.2 Results from the first radar position

The results from the first radar position are shown in Figure 6-21 and Figure 6-22. In Figure 6-21 the radar is on medium pulse and the turbine echoes are displayed as approximately 600 metres in azimuth and 70 metres down range. Whilst in Figure 6-22 the radar is on long pulse and the displayed wind farm echo sizes are respectively 610 metres by 300 metres. The eastern met. mast shows clearly, but with a significantly narrower azimuth than the turbines.

It should be noted that it was very difficult, with the radar at this low height (about 6m above sea level), to detect small targets within the wind farm itself.

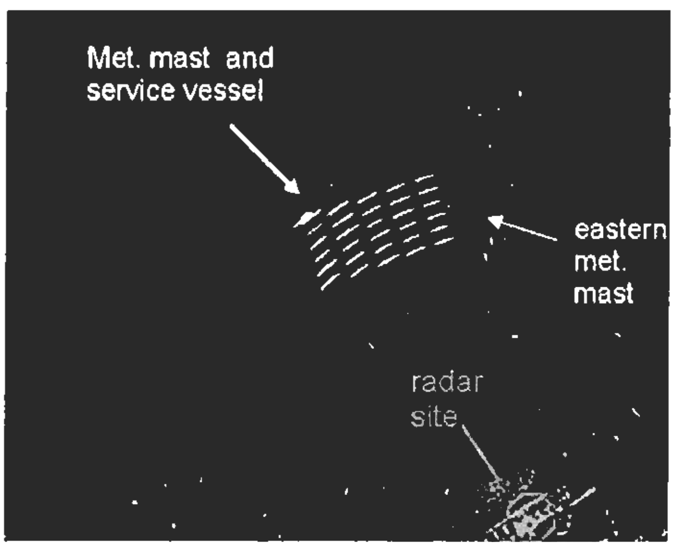



*Figure 6-21: Radar is on medium pulse*

On the medium pulse length the transmitted power was such that the eastern anemometer mast was only just detectable, but neither lifeboat could be seen on the display. On the long pulse length the turbines were very prominent, but, as with the lifeboats' own radars, the boats could only be detected rarely by the shore radar.

As with the RNLI lifeboat radars, there was no discernable variation in the magnitude of the turbine response with respect to blade disc direction or rotation. Had the blade disc direction varied to a significant extent during the trials, it might have been possible to accurately measure any variations in across range response distances.

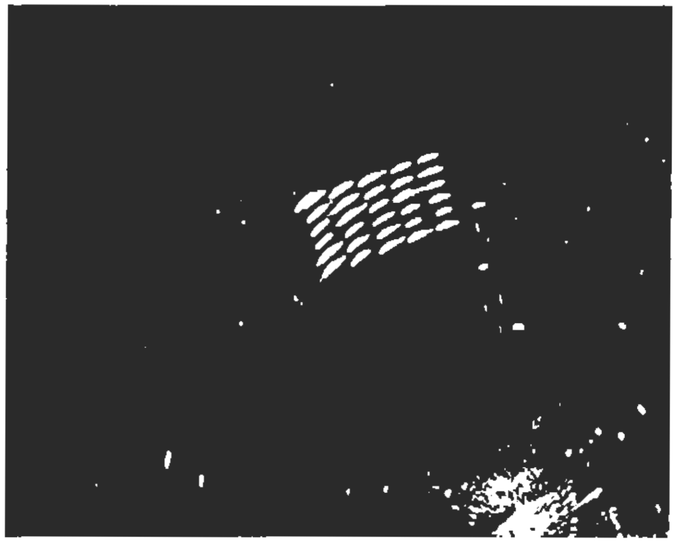
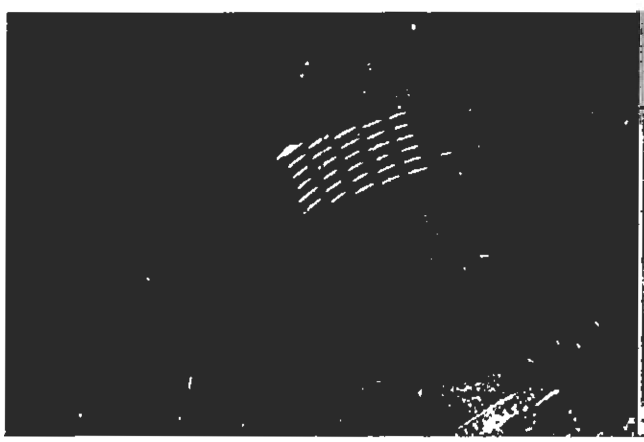


*Figure 6-22: Radar set to long pulse length*

An example of the detection of the "Lady of Hilbre" is illustrated in Figure 6-23. It can be noted in the figure that the vessel can just be detected between turbines 20 and 25.



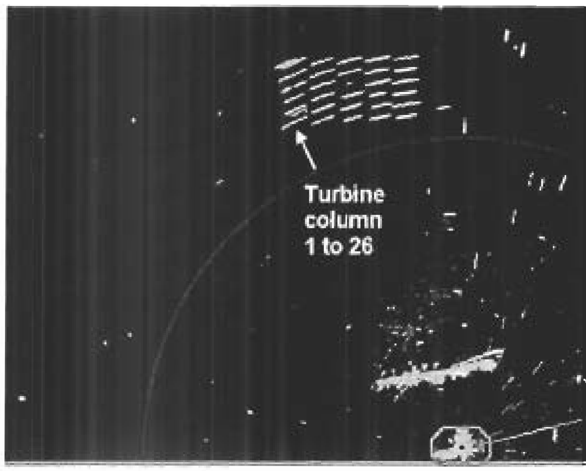
The "Lady of Hilbre" can just be detected between turbines 20 and 25

*Figure 6-23: Detection of the "Lady of Hilbre"*

## 6.13 Mobile radar at higher location

The mobile radar was then taken close to the BHP Billiton radar site at Gwaenysgor. At this site it was approximately 200 metres above sea level and 5.2 nm from the wind farm.

In Figure 6-24 it can be seen that the detection of small targets was not greatly improved but the discrimination of the western meteorology mast from turbine 26 and the service vessel immediately south of turbine 6 was apparent.




New mobile radar site (close to BHP Billiton site)

*Figure 6-24: Mobile radar at Gwaenysgor*

## 6.14 BHP Billiton radar

The radar recording unit was then transferred to the BHP Billiton Raytheon radar unit, close by. The position of this radar relative to the wind farm is shown in Figure 6-25. The displayed sizes of the North Hoyle WTGs from Gwaenysgor appear significantly greater than theoretical calculated size, the across range echo size being around 610 metres at a range of 5.2 nm and the down range depth being around 200 metres. For ARPA or VTS / Port radar tracking systems the effects may be that tracking vessels within or close to wind farms may be problematic . This was found to be the case with the "Norbay" ARPA systems and with the BHP Billiton tracking system at Gwaenysgor.

The raw radar image with high persistence level is shown in Figure 6-26. Using a high persistence level the recorded data would, when filtered, detect targets if not directly behind turbines. This is illustrated in Figure 6-27. When target vessel to the North of the wind farm was clear by approximately 1500 metres, its response was increased noticeably, as is shown in Figure 6-28.

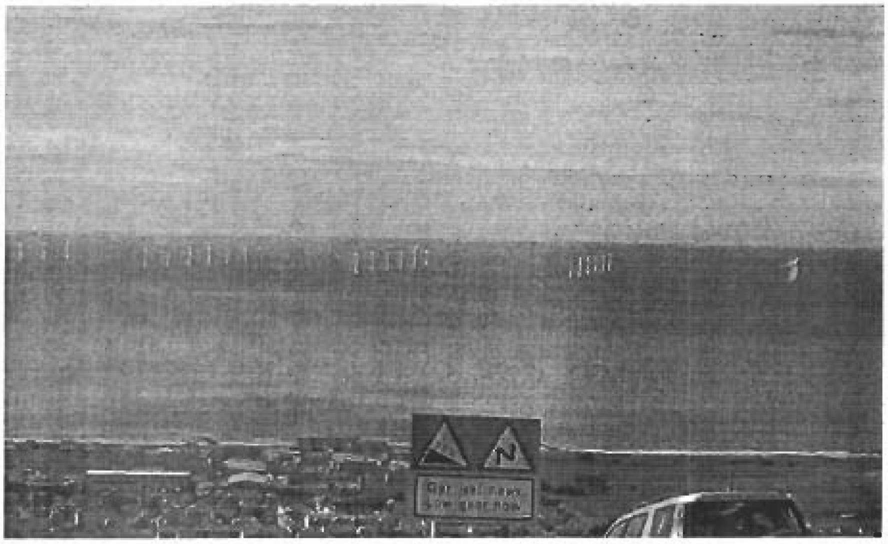


*Figure 6-25: Relative Position of BHP Billiton Raytheon radar head at Gwaenysgor*

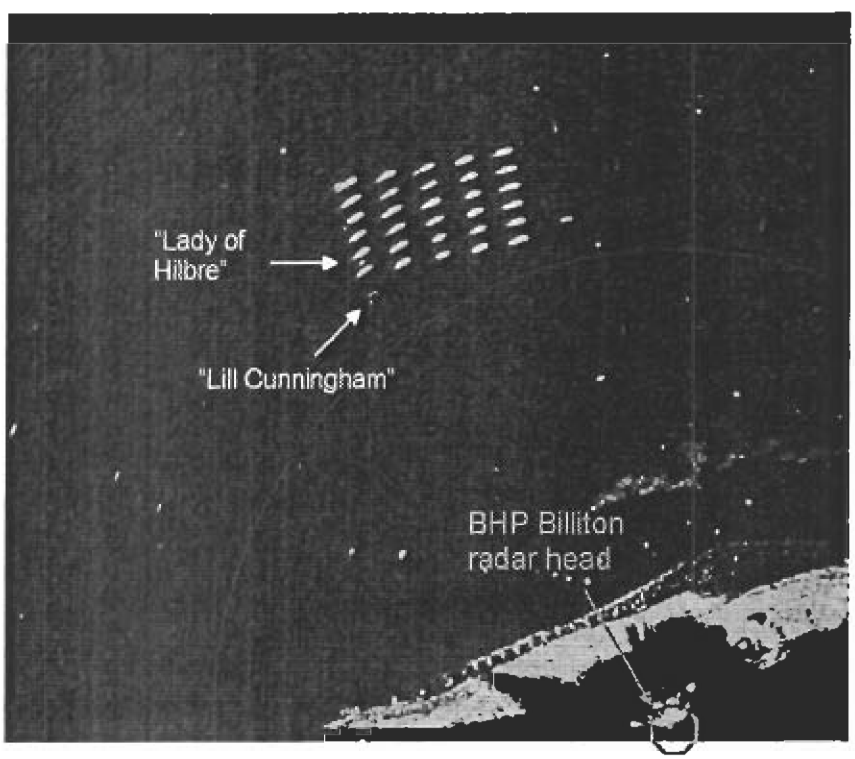



*Figure 6-26: Raw radar with high persistence level*

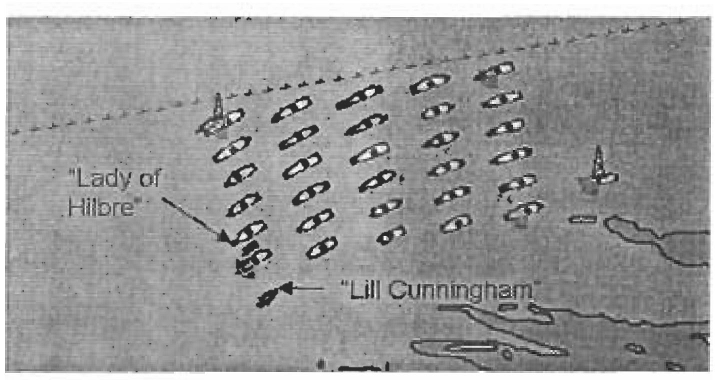



*Figure 6-27: Filtered display - high persistence*

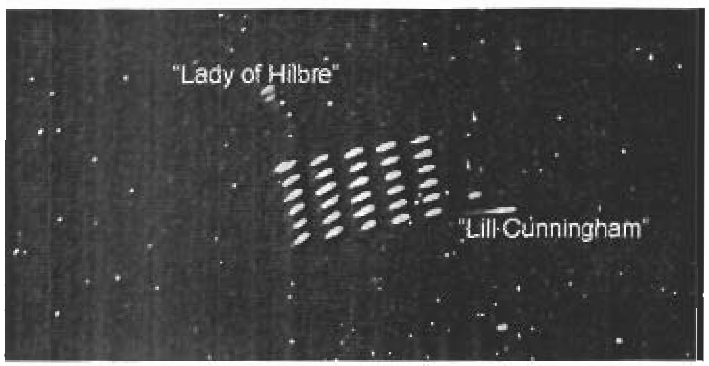



*Figure 6-28: Target lifeboats clear of the wind farm*

## 6.15 MCA larger vessel radar detection and ARPA evaluation

### 6.15.1 Overview

To evaluate the effects of wind farm structures on type-tested radars using larger scanner sizes.

The equipment required for this trial was:

- Larger vessel, with type-tested MCA approved radar equipment;
- Smaller vessel fitted with a radar reflector, carrying out a detection exercise described in the following paragraphs.

In the week following the trials undertaken by the two lifeboats, on July 29th 2004, the P & O passenger / cargo ferry MV "Norbay" was used to make a passage around and through the wind farm. During this time her officers observed the wind farm service vessel "Fast Cat" which was carrying out the detection exercise through the wind farm. The "Norbay" was herself monitored by the Mersey Docks and Harbour Board port radar, sited at Seaforth Dock, Liverpool and by the BHP Billiton radar at Gwaenysgor. The courses followed during the trial are shown in Figure 6-29.

"Norbay" was fitted with Raytheon X and S-band radars, each with Raytheon M34 Automatic Radar Plotting Aids (ARPA). "Fast Cat" was fitted with a Firdell Blipper 210-7 radar reflector.

"Norbay" also monitored her communications systems, her Automatic Identification System (AIS) and her Global Positioning System (GPS) equipment whilst within and close to the wind farm (see the Masters exercise report in sub-subsection 6.16.3).

"Norbay" has a length overall of 166.7 metres, beam 23.4 metres and 17,464 Gross Tonnage. Two photographs of the "Norbay" can be seen in Figure 6-30. Whilst a photograph of the "Fast Cat" and its radar reflector are shown in Figure 6-31.

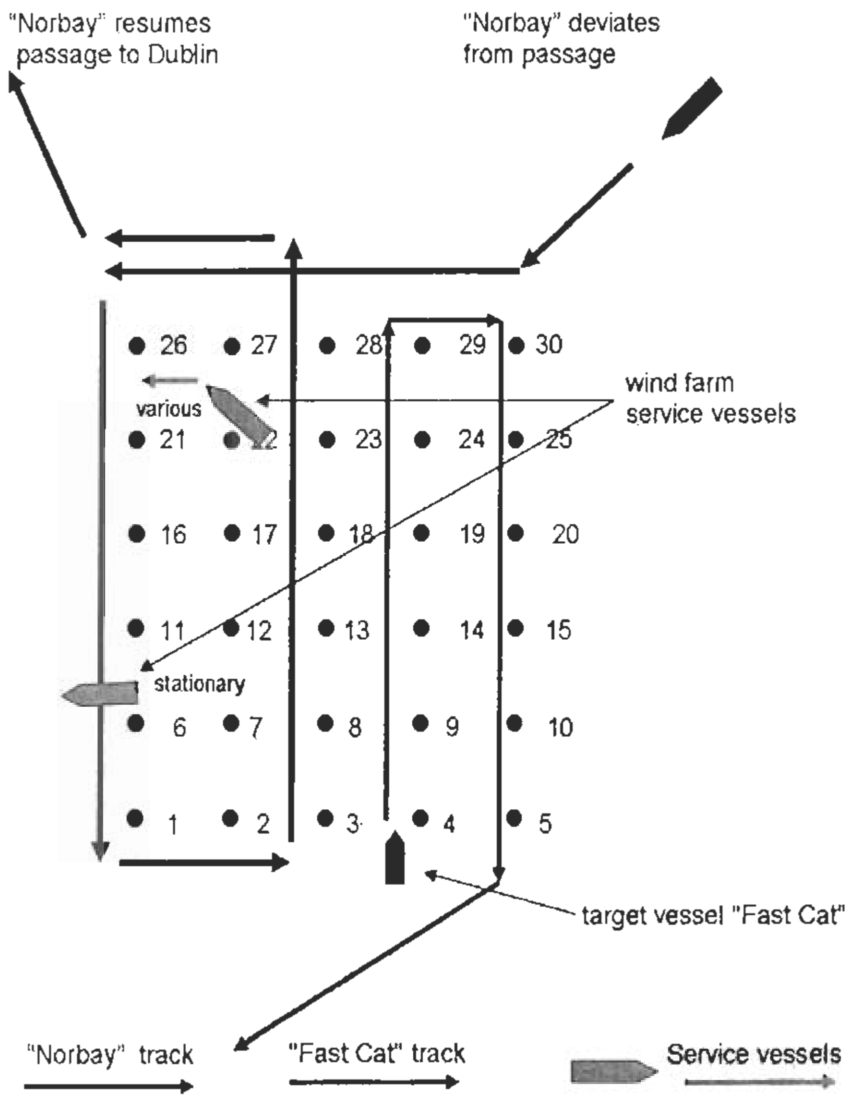



*Figure 6-29: Larger vessel trials schematic*


*Figure 6-30: MV "Norbay"*


*Figure 6-31: "Fast Cat" and its "Blipper" radar reflector*

## 6.16 Results of the Trials

The results are presented in Figure 6-32 to Figure 6-37. In Figure 6-32 the raw radar display as "Norbay" begins to pass at a distance of 800 metres across the northern boundary of the wind farm is shown. Whilst in Figure 6-33 the filtered recording of "Norbay" passing turbine 30 is presented. Note that in both the raw and processed radar displays, strong multiple echoes of turbines are visible.

As the "Norbay" passes turbine 29 multiple echoes are still visible as is shown in Figure 6-34 and in Figure 6-35 as the vessel passes turbine 28. In Figure 6-36 the raw radar display, as the "Norbay" rounds NW corner of the wind farm, shows heavy multiple and reflected echoes.

In Figure 6-37 the filtered display, with high persistence is shown. As the "Norbay" leaves the wind farm it resumes its passage with a hull aspect of about 150 degrees. No multiple echoes are see at this aspect, but some small reflected echoes are visible.

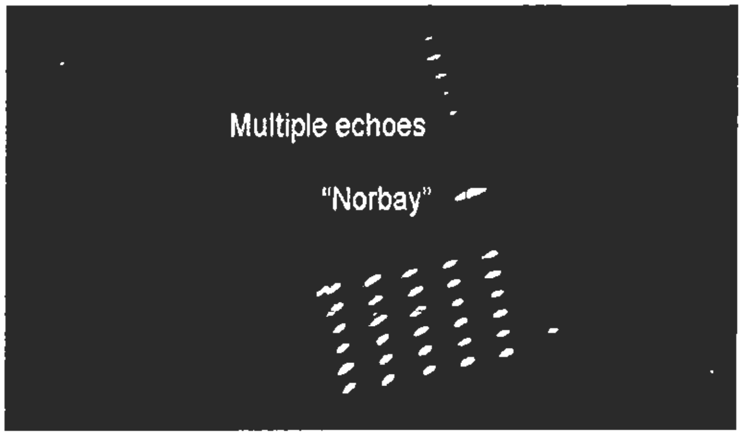



*Figure 6-32: Raw radar data as the "Norbay" passes turbine 30 at a range of 800 metres*

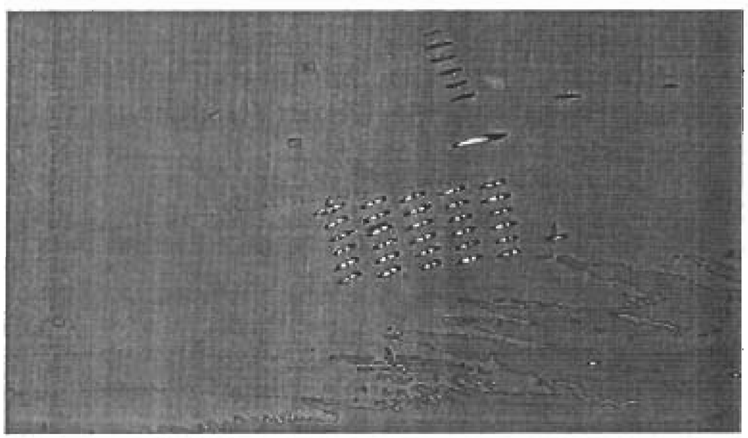


*Figure 6-33: Filtered radar data as the "Norbay" passes turbine 30 at a range of 800 metres*

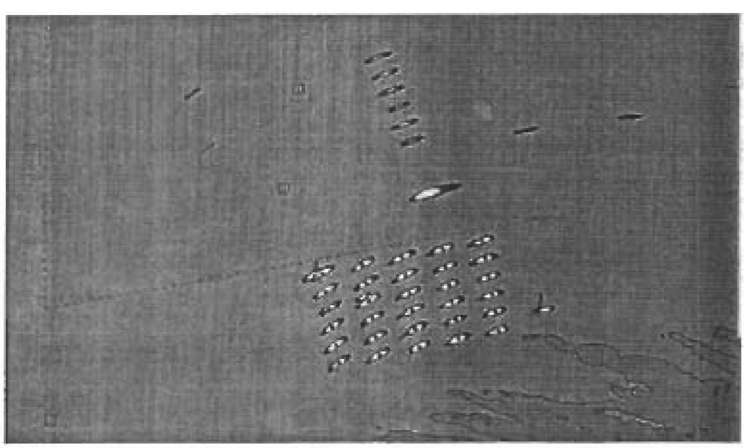


*Figure 6-34: Filtered radar data as the "Norbay" passes turbine 29*

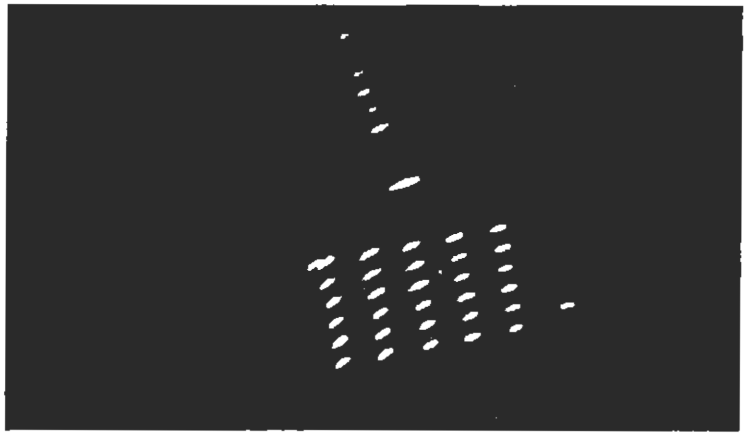


*Figure 6-35: Raw radar data as the "Norbay" passes turbine 28*

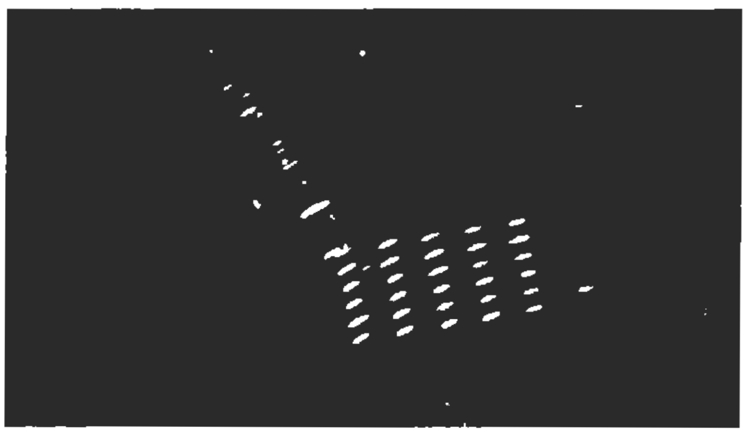


*Figure 6-36: Raw radar data as the "Norbay" rounds NW corner of the wind farm*

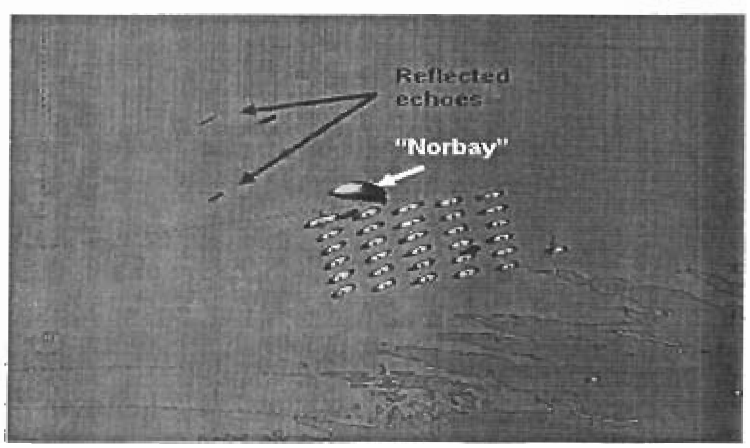



*Figure 6-37: Filtered radar data as the "Norbay" leaves the wind farm*

### 6.16.1 The Mersey Docks and Harbour Board long range radar

This radar, at a range of 14 nautical miles (26 km) from the wind farm, successfully tracked the "Norbay" during her passage around and through the turbine array, with the Norcontrol VOC500 tracking and recording equipment. However, no smaller vessels could be detected or tracked at this range.

### 6.16.2 Reflected and multiple echoes in general

Since the WTGs are strongly reflecting when vessels and / or shore based radars are close by they can produce significantly interfering reflected and multiple echoes.

Reflected echoes occur when signals are reflected at an angle from one structure to another and returned to the radar via the same route. The latter target will then be indicated on the display in the direction of the initial reflecting surface, and at a range equivalent to the total distance from radar to initial reflector plus the distance from it to the second surface.The target may additionally be indicated at its correct range and bearing.

This effect occurred within the wind farm when signals were reflected between WTGs. Multiple echoes occur similarly when two strongly reflecting surfaces reflect signals backwards and forwards between them, such that echoes of the latter target occur a number of times behind the initial reflecting target, the distance between each such spurious echo being that of the two targets.

This was found to occur with the BHP Billiton radar sited at Gwaenysgor, whose purpose is to monitor traffic in and around the Douglas and Hamilton oil and gas fields. These fields lie 14 nm from the radar site, the North Hoyle wind farm lying in the same direction but only 5.2 nm from the radar site. The Gwaenysgor radar scanner is 200 metres above sea level.

When the P & O ferry "Norbay" was proceeding along the northern boundary of the wind farm and at a distance of around 800 metres from it (as indicated by the radar ranges) very strong multiple echoes were found to occur on its far side (see subsection 6.16) At this time the "Norbay" was almost broadside on to the scanner direction, such that its reflected echoes to the WTGs would be maximum.

Both of these effects may have implications for port approaches, Vessel Traffic Services, search and rescue, and for collision avoidance. As with side lobe echoes, the effects can be reduced by turning down the receiver gain, but again with the penalty of reducing the displayed response of other vessels or buoyage.

For radars used in Vessel Traffic Services, for monitoring infringements, or in port approaches the effects of multiple and reflected echoes may be significant, particularly where a number of vessels may be required to pass or anchor close to a wind farm boundary. However, they may be reduced by the careful siting of shore radars relative to shipping routes and wind farms, or if necessary, by using radars at different sites to resolve ambiguities.

Previous laboratory studies have indicated that there is high potential for such reflected signals to trigger Racons when a turbine is within 1000 metres of them. No Racons were at this distance from the North Hoyle turbines and therefore this could not be substantiated. However, if Racons were to be considered for use in marking wind farms, this effect should be determined. Trinity House Lighthouse Service, which maintains a number of Racons, have agreed to investigate this.

### 6.16.3 Report from the Master, MV "Norbay"

**mv.NORBAY**

MCA RESEARCH INTO CLOSE NAVIGATION AROUND
THE NORTH HOYLE WIND FARM.

**Vessel's route :** West along North edge of wind farm approx. 300m off line of turbines, South along Western edge approx. 300m off line of turbines then East to midway between turbines 2 and 3 then North between rows of turbines to resume passage to Dublin.

**Weather on scene :** Light winds, strong ebb tide fine and clear.

| | | |
|---|---|---|
| **Bridge team :** | Master | M. Ingham |
| | Rel. Master | J. Moore |
| | Ch. Officer | D. McAuley |
| | 2nd Officer | A. Saulnier |

**Radar Types :** 1 x Raytheon M34 Arpa 3cm
1 x Raytheon M34 Arpa 10cm

**Observations :** Internal and external radio communications satisfactory.
AIS fully satisfactory.
All navigational equipment functioned satisfactorily.

**Radar observations :**

1. On long pulse experienced no definition between close targets.
2. Definition on 3cm radar better than the 10cm set.
3. Experienced difficulties in plotting targets running close to turbines as target swap to larger echo (turbine) occurred before plot had been calculated.
4. Small targets could only be identified when they were at a distance of more than 300m off the turbines.
5. Experienced numerous false echoes close to the turbines when about 1.5 miles off.
6. Echoes of targets on 10cm radar joined up in sweep at a distance of 0.6 miles off.
7. When vessel and targets running N/S along columns of turbines there were no problems experienced in plotting targets with both 3 and 10cm sets so long as the targets remained over 300m from turbines. However, the strength of the echo on the 3cm set faded the closer the target became.

# 7 MCA navigation system trials

## 7.1 The Global Positioning System (GPS)

Basic GPS operated satisfactorily in all areas near to and within the wind farm with no change in signal to noise ratios , indicating that there was no interference being caused to the UHF satellite signals by the wind farm generators.

The lifeboat crew did report that the Magnavox "Professional" receiver used in the "Lill Cunningham" would not accept Differential GPS signals whilst in the wind farm. The differential transmitter used in this area is sited at Point Lynas, Anglesey, using the low frequency of 297.5 KHz.

However, no other vessel has reported difficulties with the reception of Differential signals and theory suggests that wind farm structures should not affect them. Other vessels have been asked to report any failures.

## 7.2 Magnetic compasses

No problems with respect to magnetic compasses were reported. However, small vessels with simple magnetic steering and hand bearing compasses should be wary of using these close to WTGs - as of course with any structure in which there is a large amount of ferrous material.

Note : Under the DTI Renewable Energy Fund projects to be undertaken on offshore wind farms and other offshore renewable energy installation (OREI) proposals, the magnitudes and frequencies of electromagnetic and acoustic emissions from such installations will be monitored. These data could also be used to infrom navigational and other off-shore concerns.

## 7.3 Loran C Trial

### 7.3.1 Trial overview and objectives

The objectives of this trial were to see whether the wind farm structures would affect low frequency signals in general and degrade the use of Loran C equipment in their vicinity.

Since none of the participating vessels carried Loran C, portable equipment was obtained from Trinity House Lighthouse Service and set up on the "Lill Cunningham". A photograph of the Loran C receiver is shown in Figure 7-1 below.

The equipment was set up before entering the wind farm and, during exercises within the farm, connection with various chains was attempted.


MCA and QinetiQ proprietary

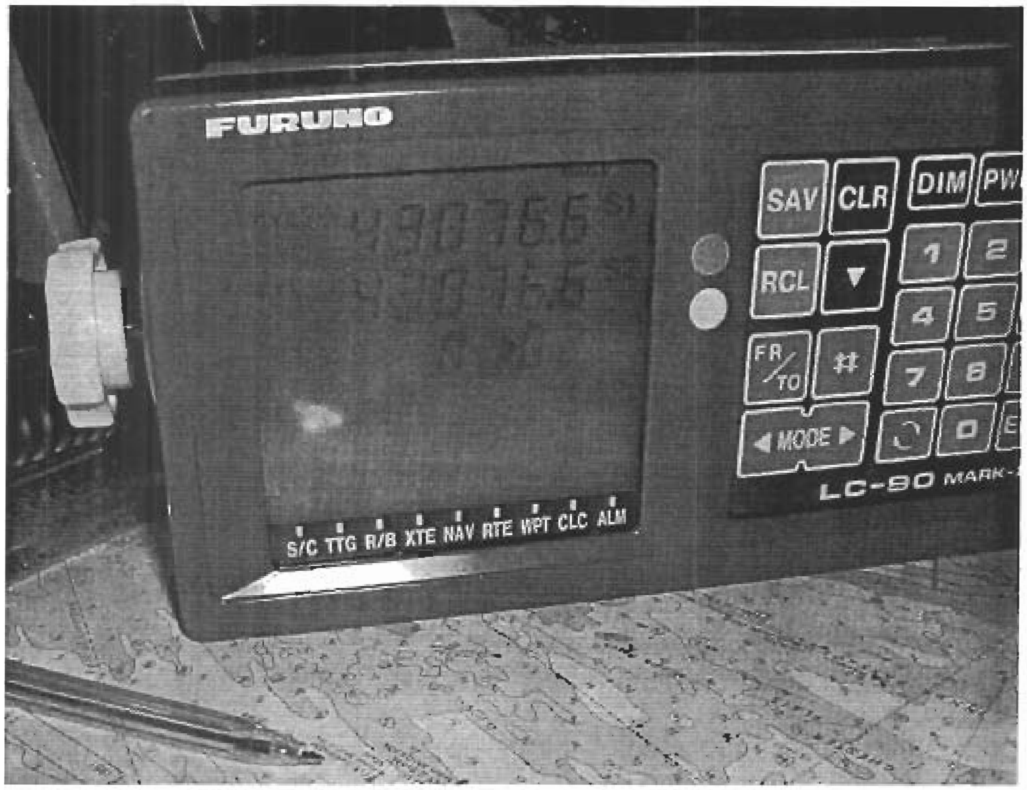



*Figure 7-1: Furuno LC- 90 Loran C receiver*

### 7.3.2 Results of the trial

Loran C, which operates at Low Frequency (100 KHz), is - currently at least - the electronic navigation fall back system if GPS were to fail. It was not fitted in any of the vessels used in the trials - being mostly used in ships on and near the US coast, although some GPS receivers have built-in Loran C software - and therefore a carry-aboard Foruno LC -90 system was used in the "Lill Cunningham".

The system failed to operate successfully and could only lock on to the Lessay Chain transmissions. Even here, only one hyperbola could be obtained. This was, however, probably due to operational errors or the closing down of the Loop Head transmitter in the Republic of Ireland, rather than the effects of the wind farm on the received signals.

The signals received jittered as would normally be expected from ground and skywave interference.

## 7.4 MCA helicopter search and rescue systems

### 7.4.1 Overview

The aim of this test was to evaluate the capabilities of search and rescue helicopters in detecting and communicating with casualties within offshore wind farms.

The following equipment was required:

- A small vessel fitted with a typical VHF radio (ideally an RNLI vessel);
- A search and rescue helicopter.

A schematic of the trial is shown in Figure 7-2. The helicopter to approach the wind farm from a direction and at a suitable height selected by its crew. The small vessel is to be positioned alongside or very close to a turbine selected by the helicopter crew, diametrically opposite the approach direction of the helicopter. The helicopter crew will attempt to detect the vessel using its radar and to communicate via VHF using a channel selected by themselves, initially when some distance away and until directly over the vessel. The helicopter crew will determine any other trials that they might wish to undertake and that might involve the use of other vessels or shore stations.

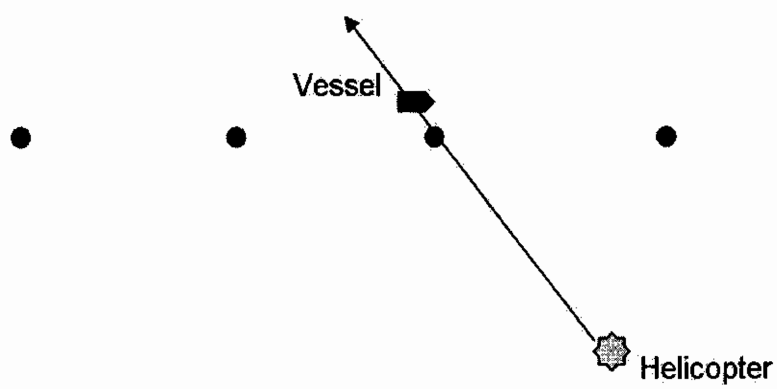



*Figure 7-2: Schematic of helicopter radar trial*

### 7.4.2 Results of the trial

There are no trials results as yet. During the original trials period, arrangements were made on three occasions for these to take place. Unfortunately on each occasion the helicopter was called out to other emergency duties and therefore the trials were cancelled.

The Commanding Officer of RAF Valley SAR Flight is keen that the trials should take place and will arrange for this with the Rhyl Lifeboat crew on a mutually convenient date. HM Coastguard Holyhead MRSC will co-operate in setting up these trails.

## 7.5 Effects of wind farm structures on non type tested radar, communications and navigation equipment

The effects on the majority of recreational vessels and their radar, communications and navigation systems will be similar to those described in the foregoing, but some non type tested systems could be more adversely affected.

During the short period of the MCA trials at North Hoyle no recreational craft were available to take part. However, the Royal Yachting Association (RYA) has asked its members to report any significant data. The letter is shown in Appendix A.

# 8 Conclusions and recommendations

## 8.1 MCA trials

MCA's programme was intended to assess the effect of the wind farm structures on marine systems in operational scenarios. The trials assessed all practical communications systems used at sea and with links to shore stations, shipborne and shore-based radar, position fixing systems, and the Automatic Identification System (AIS). The tests included basic navigational equipment such as magnetic compasses.

The effects on the majority of systems tested by the MCA were not found to be significant enough to affect navigational efficiency or safety, and an on-going collection of data on such systems is expected to prove these conclusions.

Some reported effects, such as those on short range radio devices, will be further investigated as will some scenarios which could not be assessed during the trials period, such as helicopter search and rescue operations within wind farms.

The only significant cause for concern found by the MCA during the trials was the effect of wind farm structures on shipborne and shorebased radar systems. It was determined that the large vertical extent of the wind turbine generators returned radar responses strong enough to produce interfering side lobe, multiple and reflected echoes. While reducing receiver amplification (gain) would enable individual turbines to be clearly identified from the side lobes - and hence limit the potential of collisions with them - its effect would also be to reduce the amplitude of other received signals such that small vessels, buoys, etc., might not be detectable within or close to the wind farm. Bearing discrimination was also reduced by the magnitude of the response and hence the cross range size of displayed echoes. If on passage close to a wind farm boundary or within the wind farm itself, this could in some circumstances affect a vessel's ability to fully comply with Rules 6, 7 and 19 of the International Regulations for the Prevention of Collisions at Sea and might also affect the performance of its automatic radar plotting aid (ARPA).

With respect to the multiple and reflected echoes produced when wind farm structures lie between the observing radar and a relatively high sided vessel, gain reduction will have similar effects to those described above. If, as in the trial undertaken, a shore or platform based radar is intended to detect and track traffic in port approaches, Vessel Traffic Systems or in the proximity of offshore oil or gas installations, the effects could be significant.

Recommendations from these trials are that:

- This report should be made feely available to all interested parties.
- Information appropriate to the safety of life at sea, such as recommendations with respect to navigating or carrying out activities such as fishing within or close to wind farms, should be promulgated as necessary by the use of Marine Guidance Notes, Marine Information Notes, Merchant Shipping Notices, etc.
- the siting of wind farm boundaries from recognised marine traffic routes should be determined in consultation with MCA HQ and other stakeholders using a

recommended risk assessment methodology, prior to the submission of consent applications.

- Similarly the location and relocation of fixed radar surveillance systems should be determined in consultation with relevant organisations.
- Further work to be done, as for example identified in the report with respect to adverse weather conditions, helicopter search and rescue operations, short range radio systems, non type-tested systems, etc., should be carried out as soon as practical.
- The results of such research should be promulgated where significant.
- The collation of data with respect to all offshore renewable energy installations (OREI) should be an ongoing activity.

## 8.2 QinetiQ trials

Four trials, covering the areas of GPS, VHF communications and radar tracking and radar clutter were performed by QinetiQ.

The QinetiQ GPS trial involved traversing previously defined courses through and around the wind farm. Along each course, the number of satellites visible to two different GPS systems (a Garmin 152 and a Garmin GPSIII) and the position of the ship were recorded. Our results show that on average between 8 and 11 satellites were visible at any one time providing accurate positioning to within 5 metres. The effect of wind turbines on VHF communications was investigated by QinetiQ using a hand-held VHF transceiver that was run in series with an adjustable attenuator. A link margin of 1 dB was achieved in free-space (away from any turbines). This required an attenuation of 16dB to be added to the transceiver.

To explore the shadow region behind the wind turbines, four link margins, 2dB, 3dB, 4dB and 5dB were used. These link margins correspond to a total attenuation of 15dB, 14dB, 13dB and 12dB added to the transceiver. The closest approach to turbine 21 was 500 metres and approximately 5m behind turbine 26. As expected the depth of shadow was greater when closer to a turbine. When behind turbine 21 the shadow was found to be approximately 2dB to 3dB lower than the attenuation needed to give a 1dB link margin in free space. For turbine 26 the shadow was deeper due to the closer proximity of the VHF system. It was found that behind turbine 26 the depth of shadow was approximately 10dB below the link margin in free space. The shadow depths are shallower than predicted theoretically confirming the worst case expectations of the theoretical work.

The QinetiQ radar shadowing trials provided very little evidence that shadowing of targets would present any significant problems. In particular the shadowing observed was, like the VHF trials, less than predicted in the theoretical study. Clutter in the radar display due to the presence of wind turbines was found to be quite considerable. Both ring-around and false plots (side lobe and spurious echoes) were observed. The observed problems could be suppressed successfully by using the gain and range settings of the radar. However, this may have the unwanted side-effect of no longer being able to detect some small targets.

## 8.3 Summary

Most of the effects of offshore wind farm structures on the operational use of marine radar, communications and navigation systems do not significantly compromise marine navigation or safety. Where there are questions about specific systems they will continue to be monitored and assessed when possible.

There are however some concerns about the use of both shipborne and shorebased radar in the proximity of wind farms. Wind farm structures generally have high vertical extents and therefore will return very strong responses when observing radars are close. The magnitude of such responses will vary according to transmitted radar power and proximity to the structures but may affect both the visual detection of targets and the effective operation of automatic radar plotting aids (ARPA).

These effects can be mitigated by vessels keeping well clear of wind farms in open water or, where navigation is restricted, keeping the wind farm boundaries at suitable distances from established traffic routes, port approaches, routing schemes, etc.

With respect to shorebased or offshore platform based systems, the careful siting of radar scanners in relation to traffic routes and wind farm configurations should enable any degrading effects to be minimised.

The overall results are summarised as:

i Global Positioning System (GPS)

No problems with basic GPS reception or positional accuracy were reported during the trials.

ii Magnetic compasses

The wind farm generators and their cabling, interturbine and onshore, did not cause any compass deviation during the MCA trials. As with any ferrous metal structure, however, caution should be exercised when using magnetic compasses close to turbine towers.

iii Loran C

Although a position could not be obtained using Loran C in the wind farm area, the available signals were received without apparent degradation.

iv Helicopter radar and communications systems

These trials were not carried out due to helicopter call-outs to emergencies on the trial days. The emergency services are keen that they should be undertaken when convenient with the co-operation of HM Coastguard Holyhead MRSC.

v VHF and other communications

The wind farm structures had no noticeable effects on any voice communications system, vessel to vessel or vessel to shore station. These included shipborne, shorebased and hand held VHF transceivers and mobile telephones. Digital selective calling (DSC)was also satisfactorily tested. The VHF Direction Finding equipment carried in the lifeboats did not function correctly when very close to turbines and the BHP telemetry link was similarly reported to suffer interruptions.

vi The Automatic Identification System (AIS) carried aboard MV "Norbay" and monitored by HM Coastguard MRSC Liverpool was fully operational.

vii Small Vessel radar performance.

1. The wind turbine generators (WTG) produced blind and shadow areas in which other turbines and vessels could not be detected unless the observing vessel was moving.
2. Detection of targets within the wind farm was also reduced by the cross and down-range responses from the WTGs which limited range and bearing discrimination.
3. The large displayed echoes of WTGs were due to the vertical extent of the turbine structures.
4. These returned strong responses from sectors of the main beam outside the half power (-3dB) points and the side lobes outside 10° from the main beam.
5. Although such spurious echo effects can be limited to some extent by reducing receiver amplification (gain) this will also reduce the amplification of other targets, perhaps below their display threshold levels.
6. Sea and rain clutter will present further difficulties to target detection within and close to wind farms. Weather conditions at the time of the trials were such that these effects could not be examined.

viii Shore based radar performance

1. Short range performance (less than 6 nm)
   When a small shore based radar was sited such that the height of its antenna was about six metres above sea level, its performance with respect to small vessels was similar to that of the vessel-mounted systems in terms of range and bearing discrimination and target detection within the wind farm.

   When moved to a height of 200 metres above sea level there was an improvement in range discrimination.

   When the higher powered and narrower beam width BHP Billiton radar was used, at the same height, the visual detection of targets within, and beyond, the wind farm was again improved.
2. Larger vessel detection
   A larger vessel was easily detected within and beyond the wind farm. However, while it was broadside on to the direction of the shore radar, reflections from the turbines produced strong multiple echoes. At an oblique aspect to the radar, multiple echoes did not occur, but some reflected echoes were observed.
3. Long range radar (more than 12 nm)
   When the wind farm was observed at long range by the Mersey docks and Harbour Board radar the vessel was easily detected and tracked

ix Radar and ARPA carried on larger vessels

As with small vessel radars, range and bearing discrimination were affected by the response from the WTGs. Definition was less on S band radar than on X band. Numerous spurious echoes from side lobes and reflections were reported by MV "Norbay" starting at a range of about 1.5 nm. The ship's ARPA had difficulty tracking a target vessel within the wind farm due to target swop to the stronger response. This substantiated a similar report with respect to the BHP Billiton radar's own tracking system

x Non type-tested radar, communications and navigational equipment

The effects on such systems will be similar to those tested during the trials but will vary individually with respect to transmitted power, antenna performance, radar beam width, etc. The Royal Yachting Association is assisting MCA by providing ongoing information through the experiences of its membership.

# A RYA letter

**NORTH HOYLE WIND FARM**

**Assessing effects on recreational craft communications and radar?**

**PLEASE TAKE PART AND FEED BACK YOUR EXPERIENCES**

The RYA is helping the MCA in testing the impact of offshore wind turbines on communication and radar equipment. Whilst they can see the effect on high tech equipment carried on board the MCA vessels, we need to assess the effect on small craft equipment, e.g., VHF, small boat radar, etc.

We have been asked to report back to the MCA the effects on recreational equipment which can only be done by those who use the area - your involvement in this is important.

**If you are sailing past the area, please do take part.**

**Ideally we are looking for two medium size vessels (30 foot) - but reports from individual vessels will also be valuable - fitted with radar and VHF, also Loran C if available. We need the vessels to enter the wind farm area, record the display on their radar - ideally with a digital camera - test VHF communications between vessels and also with the coastguard at Holyhead.**

<u>What to do:</u>

1 **Before entering the wind farm area, please call up the Wind Farm Operations Manager, Mike Bradley (07736631513) to check whether any maintenance vessels are operating. If maintenance vessels are operating please keep 500m clear of them**

2) **Approaching the wind farm area look at the effects on your radar screen, ideally take a digital picture of them, or sketch them out. If you turn the signal down to avoid distortion of the signal, ensure you would still be able to pick up other small vessels**

30 **Before entering the wind farm area, call the Holyhead Coastguard, District Controller, Jim Paton (01407767951) and tell him what you are doing and carry out a (VHF) radio check outside the wind farm area. If you have a hand held you may also want to carry out the exercise with this too.**

4) **Once inside the wind farm area, look again at the effect on your radar screen and report as in (2)**

5) **Once inside the wind farm area, carry out a second radio check with the Coastguard.**

6) **If you are sailing with two vessels, get behind the turbines out of direct sight of one another and test radio communications with one another. You can also check to see the effects on your radar.**

7) **Please also report the type of equipment you have on board (VHF and radar), height of VHF mast, proximity to the turbines when you carried out the recordings.**

8) **Then send your findings back to Susie Tomson (Planning and Environmental Officer) at the RYA either by phone, email or post.**

   **Contact details: Susie Tomson, RYA House, Ensign Way, Hamble, Hants, SO31 4YA. Email Susie.tomson@rya.org.uk . Please call if you have any queries my direct line is 023 8060 4222.**

Please feel free to add any other comments on your experience of sailing through the area.

*THANK YOU FOR YOUR HELP AND COOPERATION*

# B Radar specifications

## B.1 Environment Agency radar (mounted in Ford Transit van)

Racal Decca Bridgemaster 250 series specifications:

| | |
|---|---|
| Magnetron peak power | 10kW |
| Frequency | 9410 MHz ± 30MHz |
| Pulse lengths / prf | 0.05 $\mu$s 1200 Hz. |
| (nominal) | 0.25 $\mu$s 1200 Hz. |
| | 1.00 $\mu$s 600 Hz |

Racal Decca antenna specifications:

| | |
|---|---|
| Aperture size | 4 ft (1.22 m.) |
| horizontal beam width | 2° (to -3 dB) |
| vertical beam width | 24° (to -3 dB) |
| sidelobes within 10° of beam | -23 dB |
| sidelobes outside 10° of beam | -30 dB |
| Polarisation | Horizontal |
| Rotation speed | 28 rpm |

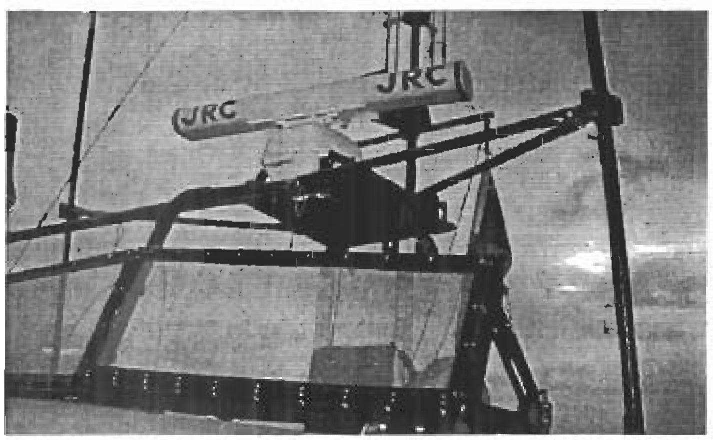
## B.2 Mersey class lifeboat radars

JRC JMA 3910 series specifications:

| | |
|---|---|
| Magnetron peak power | 10 kW |
| Frequency | 9410 MHz ± 30MHz |
| Pulse lengths (nominal) | 0.08 $\mu$s<br>0.2 $\mu$s<br>0.4 $\mu$s<br>0.8 $\mu$s |

JRC antenna specifications:

| | |
|---|---|
| Aperture size | 4 ft (1.22 m) |
| horizontal beam width | 1.9° (to -3 dB) |
| vertical beam width | 25° (to -3 dB) |
| sidelobes within 10° of beam | -23 dB |
| sidelobes outside10° of beam | -26 dB |
| Polarisation | Horizontal |
| Rotation speed | 25 rpm |



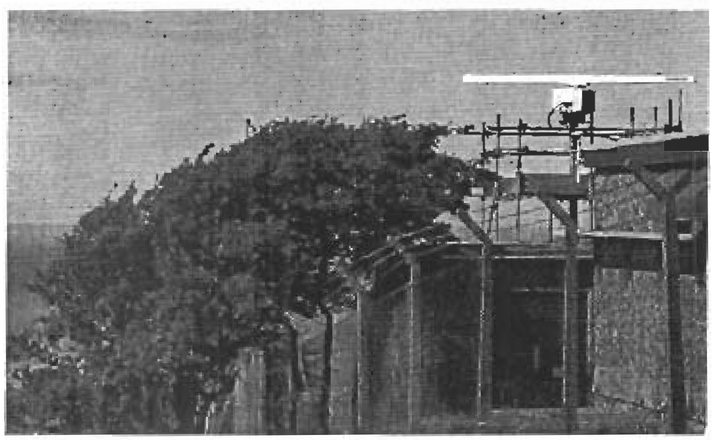
## B.3 BHP Billiton Gwaenysgor Radar (ashore above Prestatyn)

Raytheon series specifications:

| | |
|---|---|
| Magnetron peak power | 25 kW |
| Frequency | 9410 MHz $\pm$ 30MHz |
| Pulse lengths / prf | 0.06 $\mu$s 3000 Hz |
| (nominal) | 0.25 $\mu$s 2000 Hz |
| | 0.5 $\mu$s 1000 Hz |
| | 1.0 $\mu$s 750 Hz |

Raytheon antenna specifications:

| | |
|---|---|
| Aperture size | 12 ft (3.66m) |
| horizontal beam width | 0.7° (to -3dB) |
| vertical beam width | 23° (to -3dB) |
| sidelobes within 10° of beam | -30 dB |
| sidelobes outside10° of beam | -? dB |
| Polarisation | Horizontal |
| Rotation speed | 22 / 26 rpm |



## B.4 M.V. "Norbay"

Two radars, X and S band, each fitted with Raytheon M34 ARPAs

Raytheon Pathfinder specifications:

| | X band | S band |
|---|---|---|
| Magnetron peak power | 25kW | 30 kW |
| Frequency | 9410 MHz ± 30 MHz | 3050 MHz ± 30 MHz |
| Pulse lengths / prf | 0.08 $\mu$s | 0.75 $\mu$s |
| (nominal) | 0.25 $\mu$s | 1.0 $\mu$s |

Pathfinder Antennae specifications:

| | X band | S band |
|---|---|---|
| Aperture size | 7ft (2.1m) | 12 ft (3.66m) |
| horizontal beam width | 1° (to -3dB) | 1.9° (to -3dB) |
| vertical beam width | 25° (to -3dB) | 30° (to -3dB) |
| sidelobes within 10° of beam | -32 dB | -32 dB |
| sidelobes outside10° of beam | ? | ? |
| Polarisation | Horizontal | |
| Rotation speed | 22-24 rpm | 22-24 rpm |

## B.5 Mersey Docks and Harbour Board Port Radar

Uses Norcontrol VOC500 Tracking system

Decca 65160 series specifications:

| | |
|---|---|
| Magnetron peak power | 25 kW |
| Frequency | 9410 MHz ± 30MHz |
| Pulse lengths / prf | ? $\mu$s ? Hz |
| (nominal) | ? |

Decca 65276U Antenna specifications:

| | |
|---|---|
| Aperture size | 18 ft (5.49 m) |
| horizontal beam width | 0.43° (to -3dB) |
| vertical beam width | 15° (to -3dB) |
| sidelobes within 10° of beam | ? |
| Polarisation | Horizontal |
| Rotation speed | ? rpm |

# C Rules extracted from the International Regulations for Preventing Collisions at Sea

## C.1 RULE 6 Safe Speed

Every vessel shall at all times proceed at a safe speed so that she can take proper and effective action to avoid collision and be stopped within a distance appropriate to the prevailing circumstances and conditions. In determining a safe speed the following factors shall be among those taken into account:

(a) By all vessels:

- (i) the state of visibility;
- (ii) the traffic density including concentrations of fishing vessels or any other vessels;
- (iii) the manoeuvrability of the vessel with special reference to stopping distance and turning ability in the prevailing conditions;
- (iv) at night the presence of background light such as from shore lights or from back scatter of her own lights;
- (v) the state of wind, sea and current, and the proximity of navigational hazards;
- (vi) the draught in relation to the available depth of water.

(b) Additionally, by vessels with operational radar:

- (i) the characteristics, efficiency and limitations of the radar equipment;
- (ii) any constraints imposed by the radar range scale in use;
- (iii) the effect on radar detection of the sea state, weather and other sources of interference;
- (iv) the possibility that small vessels, ice and other floating objects may not be detected by radar at an adequate range;
- (v) the number, location and movement of vessels detected by radar;
- (vi) the more exact assessment of the visibility that may be possible when radar is used to determine the range of vessels or other objects in the vicinity.



MCA and QinetiQ proprietary

## C.2 RULE 7 Risk of collision

(a) Every vessel shall use all available means appropriate to the prevailing circumstances and conditions to determine if risk of collision exists. If there is any doubt such risk shall be deemed to exist.

(b) Proper use shall be made of radar equipment if fitted and operational, including long-range scanning to obtain early warning of risk of collision and radar plotting or equivalent systematic observation of detected objects.

(c) Assumptions shall not be made on the basis of scanty information, especially scanty radar information.

(d) In determining if risk of collision exists the following considerations shall be among those taken into account:

(i) such risk shall be deemed to exist if the compass bearing of an approaching vessel does not appreciably change;

(ii) such risk may sometimes exist even when an appreciable bearing change is evident, particularly when approaching a very large vessel or a tow or when approaching a vessel at close range.

## C.3 RULE 19 Conduct of vessels in restricted visibility

(a) This Rule applies to vessels not in sight of one another when navigating or near an area of restricted visibility.

(b) Every vessel shall proceed at a safe speed adapted to the prevailing circumstances and conditions of restricted visibility. A power-driven vessel will have her engines ready for immediate manoeuvre.

(c) Every vessel shall have due regard to the prevailing circumstances and conditions of restricted visibility when complying with the Rules of Section I of this Part.

(d) A vessel which detects by radar alone the presence of another vessel shall determine if a close-quarters situation is developing and/or risk of collision exists. If so, she shall take avoiding action in ample time, provided that when her action consists of an alteration of course, so far as possible the following shall be avoided:

(i) an alteration of course to port for a vessel forward of the beam, other than for a vessel being overtaken;

(ii an alteration of course towards a vessel abeam or abaft the beam.

(e) Except where it has been determined that a risk of collision does not exist, every vessel which hears apparently forward of her beam the fog signal of another vessel, or which cannot avoid a close-quarters situation with another vessel forward of her beam, shall reduce her speed to the minimum at which she can be kept on her course. She shall if necessary take all her way off and in any event navigate with extreme caution until danger of collision is over.

# References

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Waterways Analysis and Management Survey

of

Nantucket Sound Main Channel

Pollock Rip Channel

and

Great Round Shoal Channel

Submitted:

10 OCT 96

Prepared by:

QM1 K.M. Keyser
USCGC BITTERSWEET (WLB-389)

Reviewed by:

LCDR D.E. Ouellette
Commanding Officer
USCGC BITTERSWEET (WLB-389)

# EXECUTIVE SUMMARY - Nantucket Sound WAMS Review

1. The WAMS Review for Nantucket Sound includes the following waterways: Pollock Rip Channel (waterway # 1244), Great Round Shoal Channel (1245), Nantucket Sound Main Channel (1246), Nantucket Sound North Side (1247), Nantucket Sound North Channel (1261) & Muskeget Channel (1265). This review was prepared in October 1996 by CGC BITTERSWEET & completed by LT Matt Stuck of CCGD1(oan) in August 1997. The original analysis was conducted by CGC RED BEECH in March 1987. There are no federal dredging projects in any of the waterways reviewed in this study.

2. Nantucket Sound is a 35 NM long body of water running E/W along the south shore of Cape Cod, Massachusetts (see enclosure (1)). Its greatest N/S dimension is approximately 20 NM and is bounded along its southern edge by Martha's Vineyard to the west and Nantucket to the east. Numerous rock and sand shoals are present in this waterway which is well known for its extremely foggy conditions year round & 2-3 knot currents. The main thoroughfare through the Sound is Nantucket Sound Main Channel. Pollock Rip Channel and Great Round Shoal Channel serve as the Sound's east entrances (northeast and southeast, respectively). Vineyard Sound is the west entrance & Muskeget Channel is the Southern entrance between Martha's Vineyard and Nantucket. This Environmentally and Navigationally Critical waterway hosts thousands of recreational vessels daily from May to October, numerous deep draft cruise ships with drafts over 30', and commercial fishing vessels & passenger ferries year round. The majority of Cape Cod and the Islands' recreational ports access Nantucket Sound resulting in extreme vessel congestion during summer months. In the event that the Cape Cod Canal is closed due to ice, fog or marine incident, Nantucket Sound is the primary route, along with Martha's Vineyard Sound, that vessels use to transit around the Cape.

3. Major changes to these waterways which have occurred since the original WAMS Study included adding (4) buoys to Pollock Rip Channel, relocating Pollock Rip LWB "PR" 2.5 NM to mark the channel entrance, adding (2) buoys to Muskeget Channel, adding (1) buoy to Nantucket Sound North Channel and discontinuing Nantucket Sound LBB "NN".

4. There are no noteworthy discrepancy trends for aids in these waterways.

5. The current AtoN systems for all the waterways of Nantucket Sound are deemed adequate by the majority of commercial survey respondents and interviewees, and Coast Guard personnel with the following exceptions:

    a) approaching from the west, the entrance to Pollock Rip Channel is difficult to detect (particularly at night, in low visibility, and in heavy seas). This situation would be greatly eased by lighting Pollock Rip Channel BY '9';

    b) mid-channel safe water buoys 'NW' and 'NS' were cited by some mariners as more a hindrance than a help and may be candidates for discontinuation;

    c) to mark the shoal water to the S of the main channel, establishing LBB '21A' in PA 41-26.91N, 070-25.20W would also mark the safe water for mariners entering and exiting Muskeget Channel by being gated with Muskeget Channel BY '6';

These possibilities will be proposed in an upcoming Notice to Mariners to solicit more specific public comment.

TABLE OF CONTENTS

# I. INTRODUCTION

1. This is a survey of Nantucket Sound Main Channel to include Pollock Rip Channel and Great Round Shoal Channel. The survey and analysis were conducted during the period August-October 1996 by personnel from USCGC BITTERSWEET.

   A Local Notice to Mariners was published during August and September 1996 requesting comments. Phone conversations were held with all identified user groups, requesting their input in the form of a survey. Responses are enclosed. A ride along was conducted with the Steamship Authority to elicit their input.

   Nantucket Sound lies within the Group Woods Hole area of responsibility. Aids to navigation in the Main Channel are maintained by USCGC BITTERSWEET with ANT Woods Hole as secondary servicing unit.

   Nantucket Sound Main Channel, Pollock Rip Channel, and Great Round Shoal Channel are not improved waterways. However, there are numerous harbor channels leading from these waterways which are maintained by the Army Corps of Engineers. These waterways are not covered in this survey.

## II. WATERWAY DESCRIPTION

1. Nantucket Sound is bounded on the north by Cape Cod and on the south by Martha's Vineyard and Nantucket Island. It joins with Vineyard Sound to the west at a line between Nobska Point and West Chop. Together with Vineyard Sound, Nantucket Sound affords an inside route from New York to Boston, access to the ports on the south shore of Cape Cod, and to the ports on Nantucket and Martha's Vineyard. The controlling depth of Nantucket Sound Main Channel is 30 ft. The numerous shoals are adequately marked to allow for safe navigation. The bottom is generally mud and sand with the many shoals being composed of mostly hard sand. This area is known for frequent fog. Currents are around 3 knots at the western end of Nantucket Sound, with velocities reaching 2 knots in Pollock Rip Channel, and 1½ knots in Great Round Shoal Channel. Numerous ports are located along Nantucket Sound, including Falmouth, Hyannis, Harwichport and Chatham to the north, and Oak Bluffs, Edgartown, and Nantucket to the south. Anchorages are located to the north and south along the entire length of Nantucket Sound Main Channel. Traffic is year round, with the heaviest concentrations of vessels occurring in the summer around the entrances to Oak Bluffs, Vineyard Haven, Nantucket, Hyannis and Falmouth. Year round traffic is mostly passenger ferries and fishing vessels. In the summer, users include numerous pleasure boaters, cruise ships, and ferry traffic increases markedly. The largest vessels are the ferries, cruise ships, and research vessels from NOAA Woods Hole. Chart 13237, Nantucket Sound and Approaches is primarily used for navigating this waterway. The current edition (36, 13 Jul 96) adequately reflects current conditions. Coast Guard stations are located in Woods Hole, Nantucket, and Chatham.

## III. EXISTING AIDS TO NAVIGATION

1. Major Lights:

At the western end of Nantucket Sound are Nobska Light, West Chop Light, and East Chop Light. Nobska Light (LLNR 15560) shows Fl W 6s with a nominal range of 12nm. A red sector shows from 263T to 289T covering Hedge Fence Shoal and L'Hommedieu Shoal. Nobska Light is equipped with a horn (2bl 30s). West Chop Light (LLNR 13775) shows Oc W 4s with a nominal range of 11nm. It also has a red sector showing from 281T to 331T covering Squash Meadow and Norton Shoals. West Chop Light's horn characteristic is 1bl 30s. East Chop Light (LLNR 13745) shows Iso G 6s with a nominal range of 9nm. East Chop Light has no sound signal. Cape Poge Light (LLNR 13715) is located on the northeasternmost point of Chappaquiddick island. It shows a Fl W 6s with a nominal range of 9nm. The next visible light to mariners transiting Nantucket Sound Main Channel is Nantucket (Great Point) Light (LLNR 545) Fl W 5s with a 12nm nominal range. Nantucket Light has a red sector from 084T to 106T covering Cross Rip and Tuckernuck Shoals. On the southeastern tip of Nantucket is Sankaty Head Light (LLNR 555), visible to mariners transiting Great Round Shoals Channel. Sankaty Light shows Fl W 7.5s with a nominal range of 24nm. To the north Pollock Rip Channel has only one major light. It is Chatham Light (LLNR 525) Fl(2) W 10s with a nominal range of 24nm.

2. Minor Aids (Pollock Rip Channel)

Pollock Rip Entrance LWB "PR" Mo (A) (LLNR 535) (8X26 LWR) marks the entrance to Pollock Rip Channel. Pollock Rip CH BY "2A" (LLNR 13530) (1CR) marks the eastern edge of Bearse Shoal for vessels transiting south from Chatham to enter Pollock Rip Channel. Pollock Rip CH LGB "4" Fl R 4s (LLNR 13535) (8X26 LGR) marks the northern edge of the channel and a 12 ft spot to the north of the channel. Channel BY "5" (LLNR 13540) (1CR) marks the south edge of the channel and a 11 foot shoal. LBB "6" Fl R 6s (LLNR 13545) (8X26 LBR) marks the north edge of the channel and a 9 ft spot. LB "8" FL R 6s (LLNR 13550) (8X26 LR) marks the outside of the turn to the southwest, and the southeastern edge of Monomoy Point. Channel BY "9" (LLNR 13555) (1CR) marks the inside of the turn to the southwest, and a 16 foot spot. LBB "10" Fl R 4s (LLNR 13565) (8X26 LBR) marks the outside of the turn to the southwest, and a 6 foot spot. Stone Horse Shoal BY "11" (LLNR 13570) (2CR) marks the western edge of Stone Horse Shoal. Handkerchief Shoal BY "12" (LLNR 13575) (2NR) marks the eastern portion of Handkerchief Shoal. Handkerchief Shoal BY "14" (LLNR 13580) (1NR) marks the southern tip of Handkerchief Shoal.

3. Minor Aids (Great Round Shoal Channel)

Great Round Shoal Channel Entrance LWB "GRS" Mo(A) (LLNR 13585) (9X35 LHR) marks the entrance to Great Round Shoal Channel. Gated pair LB "1" Fl G 2.5s (LLNR 13590) (8X26 LR) and LB "2" Fl R 4s (LLNR 13595) (8X26 LR) mark the commencement of the channel from the east. Gated pair LB "3" Fl G 4s (LLNR 13600) (8X26 LR) and LB "4" Fl R 2.5s (LLNR 13605) (8X26 LR) are the next aids to the west. Next, gated pair LB "5" Fl G 2.5s (LLNR 13610) (8X26 LR) and LB "6" Fl R 4s (LLNR 13615) (8X26 LR) mark the outer edges of the channel. In addition LB "6" marks the southernmost edge point of Great Round Shoals. To facilitate the turn to the northwest, gated pair BY "9" (LLNR 13625) (1CR) and BY "8" (LLNR 13620) (1CR) mark the outside and inside of the turn respectively. LWB "GRC" Mo(A) (LLNR 13630) (8X26 LWR) serves as a mid-channel buoy between "8" and "9" to separate inbound and outbound traffic in the turn. Point Rip Shoal BY "11" (LLNR 13635) (3CR) marks shallow water to the west of the channel. LBB "13" Fl G 2.5s (LLNR 13640) (8X26 LBR) marks the inside of the turn most vessels take to the west to enter Nantucket Sound Main Channel. The final aid in Great Round Shoal Channel is LB "15" Fl G 4s (LLNR 13655) (8X26 LR) marking the western boundary between Great Round Shoal Channel and Nantucket Sound Main Channel.

4. Minor Aids (Nantucket Sound Main Channel)

The Main Channel is bounded to the north and south by shoal waters. Most aids that define the channel do so by marking dangerous shoal water. From east to west the aids are as follows: Tuckernuck Shoal LBB "1" Fl G 4s (LLNR 13660) (8X26 LBR) marking the east end of Tuckernuck Shoal, used mainly by ferries and mariners transiting south to Nantucket. Nantucket Sound Main Channel LGB "17" Fl G 6s (LLNR 13665) (8X26 LGR) marking a 17-28 foot shoal to the south. Halfmoon Shoal LBB "18" Fl R 4s (LLNR 13675) (8X26 LBR) marking Halfmoon Shoal to the north. Horseshoe Shoal LB "20" Fl R 4s (LLNR 13690) (8X26 LR) marks the southernmost portion of Horseshoe Shoal to the north. Almost directly south of LB "20" is Cross Rip Shoal LGB "21" Fl G 2.5s (LLNR 13685) (8X26 LGR) marking the shallows of Cross Rip Shoal to the south. Next is Nantucket Sound LWB "NS" Mo(A) (LLNR 13700) (8X26 LWR) acting as a midchannel buoy separating inbound and outbound traffic. Nantucket Sound Main Channel Wreck LB "20WR" Q R (LLNR 13710) (8X26 LR) marks a wreck with a least known depth of 40 feet. Hedge Fence LGB "22" Fl R 4s (LLNR 13720) (8X26 LGR) marks the southeastern portion of Hedge Fence Shoal which lies to the north of the channel. Squash Meadow East End BB Green/Red (LLNR 13725) (8X26 BR) marks the eastern portion of Squash Meadow shoal to the south. Squash Meadow West End BY Green/Red (LLNR 13730) (3CR) marks the West end of Squash Meadow to the south. East Chop Flats LBB "23" Fl G 4s (LLNR 13735) (6X20 LBR) marks the shoal waters off East Chop Martha's Vineyard to the south. Nantucket Sound West LBB "NW" (LLNR 13750) (8X26 LBR) marks the entrance to Nantucket sound from the west in Vineyard Sound. To the north of "NW" is Hedge Fence West End BY Red/Green (LLNR 13755) (2NR) marking the western end of Hedge Fence Shoal. West Chop LGB "2" Fl R 4s (LLNR 13760) (8X26 LGR), although primarily used for vessels transiting south into Vineyard Haven, also marks the shoals off West Chop, Martha's Vineyard nicely for boats travelling through Nantucket Sound.

## IV. RADIONAVIGATION AIDS

1. There are no Radiobeacons transmitting in Nantucket Sound, Pollock Rip Channel, or Great Round Shoal Channel.

2. The only Racon Equipped aid in the area is Great Round Shoal Channel Entrance LWB "GRS" showing morse code "G" ( _.. )

3. GPS, DGPS, and Loran coverage for the area is excellent. Usable DGPS beacons include Chatham (325 kHz), Montauk (293 kHz) and Portsmouth (288 kHz). Loran coverage is obtained using the 9960 W,X,Y, and Z chains. During low visibility, most mariners asked responded they used radar and GPS as their primary means of navigation.

## V. WATERWAY USERS

1. Commercial vessel users include the Woods Hole-Martha's Vineyard-Nantucket Steamship Authority, Falmouth Ferries, Hy-Line Cruises, Patriot Party Boats, American Cruise Lines, Island Commuter Corporation, and the New England commercial fishing fleet. Towing, fishing, and ferry traffic is year-round, with volume increasing substantially during the summer months. The cruise lines operate only during the summer.

2. Recreational traffic is heaviest during mid-May to October. The heaviest concentrations of traffic occur in the approaches to Nantucket, Edgartown, Oak Bluffs, Vineyard Haven, and Woods Hole harbors.

3. The Woods Hole Oceanographic Institute and NOAA have several large research vessels homeported in Woods Hole.

4. There are numerous small commercial fishing vessels homeported in the many harbors along Nantucket Sound.

## VI. SERVICING UNITS

1. Aids to Navigation in Nantucket Sound are serviced primarily by CGC BITTERSWEET and ANT Woods Hole. CGC WHITE SAGE serviced numerous aids in the area, but was recently decommissioned. This has presented problems to both the BITTERSWEET and the ANT team. Aids which are too large for the ANT boats to handle are now serviced by the BITTERSWEET. However, the draft of BITTERSWEET precludes it from entering many of the areas WHITE SAGE used to service. This has caused Bittersweet to have to relocate aids in order to be able to work them. This situation should be alleviated with the commissioning of the CGC IDA LEWIS.

## VII. VESSEL MANAGEMENT SYSTEM

1. There are no traffic separation schemes or manned regulatory systems in effect. None are recommended.

## VIII. CRITICALITY

1. The results of this survey confirm the navigational criticality of this waterway.

## IX. PUBLIC COMMENT

1. A Local Notice to Mariners regarding this WAMS was published during Aug-Sep 96. Mariners who responded were sent a survey to fill out. Six surveys were returned, and are enclosed.

2. Surveys were sent to all identified user groups to solicit their ideas for ways to improve the waterway. Responses are enclosed.

3. A summary of proposed user changes is as follows (BITTERSWEET'S reaction to these changes follows each.):

a) Relocate Tuckernuck Shoal LBB "1" (LLNR 13660) to the west from its present position. The aid had been moved steadily east in recent years, primarily due to its proximity to shoal water for the servicing unit. Recently it was moved further east to mark the position of the submerged F/V Sea Lion. This causes congestion between pleasure craft and ferries entering Nantucket Harbor. BITTERSWEET agrees with this to a point. We originally moved the aid 100 yds to the east to increase the distance to shoal water when we work the aid. This aid should be relocated to this point.

b) Light Pollock Rip Channel BY "9" (LLNR 13555). This aid is used by many who transit the channel from west of Monomoy Island. Many times its radar return is lost in the sea clutter and cannot be acquired. If the aid was lit it would provide the vessels, many of which transit at night, a better reference to line up with the rest of the channel. BITTERSWEET agrees with this proposal also. The mariner would be better served by a lighted aid in this position.

c) Concern was expressed by CDR Sutton, the Captain of NOAA Ship ALBATROSS IV, with the use of midchannel and lateral aids. He stated that many times fishing vessels and pleasure boats ignore the "keep as far as practicable to the right side of the channel". When this occurs near midchannel buoys, they interfere with the ability of larger ships to maintain their course. CDR Sutton suggested the establishment of a traffic scheme for Great Round Shoal Channel. BITTERSWEET does not feel a traffic scheme is needed. However, we do agree with the issue of the midchannel buoys. This is further discussed under our proposed changes in the waterway system design analysis section.

d) CAPT J. Gibbons of the Northeast Pilots suggested designated anchorages for the cruise ships in Vineyard Haven, Oak Bluffs, Edgartown, and Nantucket. This is an idea which should be pursued. As it stands now, cruise ships simply move out of the channel and anchor in the general anchorages. If a designated area were established for cruise ships, it would better regulate traffic congestion around these areas.

The following responses to our surveys do not pertain to the waterways addressed in this WAMS. However, we felt that they should be included because of the high traffic use of Hyannis harbor.

e) Hyannis harbor should be dredged to a controlling depth of 14 ft. The controlling depth is now 11 ft, and the draft of the ferries transiting the harbor is 10 ft. In addition, many of the F/V homeported in Hyannis have deeper drafts. A 14 ft controlling depth would be much safer for all waterway users.

f) Daniel Horn, Harbormaster for the town of Barnstable, expressed concern about the marking of SW rock. The present marking system does not adequately show the danger of SW rock and a smaller rock to the east.

g) Wayne Kurker, president of the Hyannis Marina, addressed many concerns about the aids in the Hyannis approach. These are included in his letter which is enclosed.

## X. WATERWAY SYSTEM DESIGN ANALYSIS

1. Nantucket Sound is adequately marked but the following changes could be instituted to better serve the mariner:

a) Light Pollock Rip Channel BY 9 (LLNR 13555).

b) Establish special anchorages for cruise ships in Nantucket Sound.

c) Conduct a survey of the area in the vicinity of LB "20WR" (LLNR 13710). If the depth of the wreck is indeed 40', we recommend discontinuing the aid, as the advertised controlling depth of Nantucket Sound Main Channel is 30'.

d) Discontinue midchannel buoys "NS" (LLNR 13700) and "NW" (LLNR 13750). As stated in the public comments, these are more of a hinderance than a help. In the case of "NW" the area is already well marked with aids. As for "NS" we recommend adding a green aid as discussed below.

e) Establish LBB "21A" Fl G 4s Bell in position 41 26.91N 070 25.20W. This would mark the shoal water to the south of the main channel, and would also mark the safe water for boats entering and exiting Muskeget Channel as a gated pair with R "6". In conjunction with this, we recommend discontinuing Muskeget Channel BB "7" (LLNR 15385) . This aid marks a 15' spot at the northern approach to Muskeget Channel. However, there is an 11' spot 600 yds to the east inside the channel that is not adequately marked.

## XII. ENCLOSURES

| | | |
|---|---|---|
| Encl: | (1 thru 6) | Public survey responses |
| | 7 | Chartlet of proposed change "a" |
| | 8 | Chartlet of proposed change "d" |
| | 9 | Chartlet of proposed changes "c", "d", "e" |

U.S. Department of
Homeland Security

United States
Coast Guard

Commanding Officer
U.S. Coast Guard
Marine Safety Office Providence

20 Risho Ave
East Providence, RI 02914-1208
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16670
12 July 2004

Colonel Thomas L. Koning
District Engineer
Department of the Army
New England District, Corps of Engineers
696 Virginia Road
Concord, MA 01742-2751

Dear Colonel Koning:

The Coast Guard has received substantial public interest and comment on the Navigational Risk Assessment of August 18, 2003, which was prepared for the Corps as part of the Nantucket Sound wind farm permitting process. Two documents in particular have raised issues that may need more examination and evaluation before a Draft Environmental Impact Statement is published.

1. The McGowan Group conducted a review of the Navigational Risk Assessment for The Alliance to Protect Nantucket Sound. One issue raised in the report is the impact, if any, that the Wind Generating Towers (WTGs) may have on the reception and accuracy of Geographic Positioning System (GPS) equipment, including Differential GPS. (See the "Electronic Interference" section beginning on page 26 of the McGowan Group report.) It would be helpful if the Navigational Risk Assessment addressed the impact, if any, that the proposed WTGs may have on shipboard GPS receivers, including commercial, recreational, and fishing vessels.

2. Both the McGowan Group report and a letter dated June 29, 2004, from The Woods Hole, Martha's Vineyard and Nantucket Sound Steamship Authority raise concerns about the impact WTGs may have on bottom contour. Specifically, The McGowan Group report questions how changes, if any, to the bottom contour may affect substructure stability, and the Steamship Authority questions whether WTG-induced bottom contour changes may cause uncharted shallow areas in the shipping channel. (See the "Environmental Influences and Impact" section beginning on page 27 of the McGowan Group report, and the first paragraph, second page, of the Steamship Authority letter, respectively.) A discussion of the effects, if any, that the WTGs may have on bottom contour is recommended.

3. The Steamship Authority letter addresses the possibility of large ice chunks being hurled from the WTG blades. An analysis of this phenomenon and its potential impact on navigation safety is recommended.

4. The Steamship Authority letter states that, due to adverse weather, its captains sometimes must use tacking maneuvers to ease a ferry's ride and improve safety. The Navigation Risk Assessment should address what impacts, if any, WTGs may have on this practice. The analysis of this issue should include a detailed discussion of tacking track lines actually used in the past, and the frequency and rationale of those tacking maneuvers.

16670
12 July 2004

Since the Corps is already in receipt of the McGowan Group report and the Steamship Authority letter, I have not included copies with this letter.

Thank you for your consideration. I look forward to continuing our support of the Corps in this Federal permitting process

Sincerely,

M. E. Landry

M. E. LANDRY
Captain, U.S. Coast Guard
Captain of the Port

Copy: CGD ONE (m, oan, dgp)
GRU Woods Hole

U.S. Department of Homeland Security
United States Coast Guard

Commanding Officer
U.S. Coast Guard
Marine Safety Office Providence

20 Risho Ave
East Providence, RI 02914-1208
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Email: EleBlanc@MSOProv.uscg.mil

16670
14 February 2005

Colonel Thomas L. Koning
District Engineer
Department of the Army
New England District, Corps of Engineers
696 Virginia Road
Concord, MA 01742-2751

Dear Colonel Koning:

As the Coast Guard continues to receive comments on navigation safety in connection with the proposed Nantucket Sound wind farm, we have identified several issues that require further analysis. These issues are in addition to those identified in our letter to you of 12 July 2004, which is attached. I understand the public comment period for the proposed facility ends on 24 February 2005, and once all comments have been received and cataloged, we will examine those pertaining to navigation safety and may identify additional issues requiring further analysis. However, to permit a timely review and analysis of issues we are aware of now, I request the Corps' attention to the following items:

1. Radar Interference. Page 28 of the Navigation Risk Assessment (Appendix 5.12-B to the Cape Wind Energy Draft Environmental Impact Statement (EIS)) discusses potential radar interference in the proposed Nantucket wind farm, as references radar experiences at Horns Rev wind park in the North Sea. A recently published report by the British Maritime and Coastguard Agency entitled "Results of the electromagnetic investigations and assessments of marine radar, communications and positioning systems undertaken at the North Hoyle wind farm by QinetiQ and the Maritime and Coastguard Agency" discusses radar interference experienced at the North Hoyle wind farm. The results of this study, and how it may relate to or impact the proposed Nantucket wind farm, should be included in the Final EIS. Since the Corps already has a copy of the North Hoyle study, I have not included a copy with this letter.

2. Impact Analysis: Section 4.3.3 of the Navigation Risk Assessment contains an impact analysis using the M/V *Eagle* as the largest commercial vessel that routinely operates near Horseshoe Shoal. However, the McGowan Group Report (first referenced in the attached letter) identified two vessels larger than the M/V *Eagle* that have been sighted in Nantucket Sound. I request an impact analysis similar to that done for the M/V *Eagle* also be conducted for the M/V *Clipper Adventurer* and the T/V *Great Gull*.

3. Ice Floes: Section 4.4 of the Navigational Risk Assessment discusses the potential effects of ice build up around and within the proposed Nantucket wind farm. However, in light of the severe icing experienced in January and February of 2004, there is concern that large, thick ice floes could develop in Nantucket Sound, outside of the wind farm, and drift into the wind farm where it may damage one or more towers, or otherwise affect navigation safety. I request an analysis of the potential for ice floe formation and drifting within Nantucket Sound, and the potential for ice floe damage to the towers.

16670
14 February 2005

4. Other Issues in the McGowan Group Report: There are many issues in the McGowan Group Report that have not been specifically addressed in my letter of July 12, 2004, or this letter. The report's relevant issues are succinctly listed in its table of contents. I request that the McGowan Group Report be examined in its entirety and that the issues raised in that report be specifically addressed in the Final EIS.

Finally, I request that this letter and my letter of 12 June 2004, along with the McGowan Group Report and the North Hoyle study, be included in the public docket for the Nantucket Sound wind farm proposal.

Thank you for your consideration. I look forward to continuing our support of the Corps in this Federal permitting process.

Sincerely,

M E Landry

M. E. LANDRY
Captain, U.S. Coast Guard
Captain of the Port

Enclosure

Copy: CGD ONE (m, oan, dgp)
GRU Woods Hole

# Commercial Fishing in Nantucket Sound: Considerations pertinent to the proposed wind farm on Horseshoe Shoals

Madeleine Hall-Arber, PhD, MIT Sea Grant College Program
David Bergeron, Executive Director, Massachusetts Fishermen's Partnership
Rhonda Ryznar, PhD, Massachusetts Institute of Technology

## *Introduction*

The Massachusetts Fishermen's Partnership (MFP) held a focus group in Hyannis on April 2, 2004 to hear from representatives of each of the fishing sectors that traditionally fish in Nantucket Sound. Participants were interviewed and asked to mark charts of the Sound showing customary areas of commercial fishing. Dr. Ryznar worked at a computer with the individuals to digitize the detailed information so that geographic information systems (GIS) could be used to visualize, analyze and display the fishermen's knowledge of the Sound. Later, fishermen of Provincetown convened their own meeting (without the presence of social scientists) and similarly marked charts to show where they usually fish in the Sound. Dr. Ryznar digitized the information from the charts that these fishermen provided.

The information collected at these two meetings was supplemented with landings data from Massachusetts Division of Marine Fisheries and some information gathered from other fishermen. This report is intended to convey the input of the focus group participants, and to relate their consensus of the need to warn of the potential impacts of the wind farm project. The participants feel strongly that these impacts should be systematically investigated before continuing with this project, or any project that may alter traditional economic use of coastal waters.

The term "systematically" is used intentionally. The wind farm project is being considered in isolation. The determination of whether or not the removal of mobile fishing effort from Horseshoe Shoals will ultimately be responsible for the loss of income to a few or to many is an important part of impact analysis. However, a project's impact must also be considered in the context of other changes that may result in cumulative impacts with more serious consequences than any single project or regulation. For example, federal fisheries regulations have severely limited the time fishermen are allowed to fish (days at sea) for groundfish, so access to inshore areas with species other than groundfish is significantly more important than in the past. Consequently, the loss of Nantucket Shoals area is potentially more deleterious than it would be if fishermen faced no other restrictions.

The predictions of potential impacts identified in this report are a compilation of those expressed by the focus group participants and do not necessarily reflect the opinions of the authors or of the institutions or programs with which the authors are affiliated.

## *Fishing Communities*

In the views of the fishermen interviewed, placement of the wind towers would make navigation of mobile fishing gear between the towers hazardous or impossible (see diagrams submitted by William Amuru in Appendix 1). Information collected suggests that mobile gear fishing vessels would be displaced from Woods Hole, Cotuit, Hyannis, and Provincetown. According to the Massachusetts Division of Marine Fisheries, 1,162,529 pounds of squid and fish were harvested in 2000 by mobile gear fishing vessels working in Nantucket Sound. According to the fishermen interviewed who fish in the Sound, a major portion of their catch is from Horseshoe Shoals. Loss of access to Horseshoe Shoals will certainly displace fishing effort to other areas in and near Nantucket Sound. This raises the potential for crowding, gear conflicts and habitat impacts elsewhere in the Sound, thereby affecting, albeit indirectly, additional fishermen and a broader range of fishing communities.

The representative fishermen who were interviewed identified vessels from Woods Hole, Cotuit, Hyannis, Marshfield, Sandwich, Chatham and Provincetown that regularly fish in the Sound. Many of the fishermen who work in the Sound are carrying on a multi-generational family tradition.

Two hundred eighty-nine individuals including commercial and recreational vessel owners, crew, shellfishermen, harbormasters, boat builders, and various other supporting businesses signed a petition asking that the wind farm not be built on Horseshoe Shoals.

## *Fish Species*

The primary commercial species that are sought in the Sound include:

- Bluefish (*Pomatomas saltatrix*),
- Black sea bass (*Centropristis striata*)
- Bonita (*Sarda sarda*)
- Mackerel (*Scomber scombrus*)
- Scup (*Stenotomus chrysops*)
- Striped bass (*Morone saxatilis*)
- Squid (*Loligo & Illex)*
- Summer flounder (*Paralichthys dentatus*), also known as fluke
- Tautog (*Tautoga onitis*).

In addition, shellfish such as conch (Knobbed whelk (*Busycon carica*), North Atlantic whelk, (*Buccinum undatum*) channeled or lightening whelks, quahogs (*Mercenaria mercenaria*), and lobsters (*Homarus americanus*) are also harvested in the Sound. Recreational fishermen catch striped bass and bluefish. Currently, there is no directed commercial lobstering on Horseshoe Shoals in the Sound; however, lobsters are caught and landed from large areas of the Sound by other fishermen. Furthermore, some fishermen suggest that lobsters migrate across the shoals at various times of the year. Recently, the deepest section of the Sound was found to be a breeding area for elvers.

## Gear

The gear used by commercial and recreational fishermen in the Sound include:

- Otter trawls (for squid, fluke, sea bass and scup)
- Hooks (commercial and recreational)
- Conch pots
- Fish pots
- Lobster pots
- Shellfish drags and dredging gear
- Fish weirs in shallow areas (15-20 feet)
- Gillnets- (only a few fishermen with bait licenses set 300 feet of gillnet to catch menhaden.)

## Landings and Income

Landings are limited by regulations that place a maximum quota of 500 pounds per day on the commercial groundfish vessels. Nevertheless, the fishermen who traditionally fish in the Sound estimate that 50 to 60% of their annual income is from the Horseshoe Shoals area. Before the recent regulations, the quota was 2000 pounds/day and prior to that, there was no limit. With the reductions on allowable groundfish landings and days at sea, access to Horseshoe Shoals for non-groundfish species is even more important economically than it was in the past.

Commercial fluke (summer flounder) landings were estimated to be approximately $2 million annually. Recreational fishing for fluke also has a significant economic value to the region.

Processing plants in Rhode Island and elsewhere rely on squid caught in the Sound. Though fishermen are not able to fish the Sound in the winter, the squid caught in season is more abundant than the plants can process and sell, therefore, a portion is frozen and stored so the processing plants are able to maintain a year-around operation.

The most recent data that gives insight into the quantities and values of fish harvested in Nantucket Sound are the Massachusetts Division of Marine Fisheries figures for the 2000 fishing year. (Caveat: These are preliminary figures, some quantities may change as they are verified.)

| Species | Vessels | Gear | Landings (in pounds) | |
|---|---|---|---|---|
| Squid (Loligo, Ilex & unspecified) | 34 | Trawl | 637,522 | |
| Fluke | 58 | Trawl | 508,785 | |
| | | Hand line | 63,598 | |
| | | Trap | 707 | |
| | | Scottish Seine | 100 | |
| Conch (Channeled, Knobbed, Lightening whelk and unspecified) | 17 | Trawl | 16,222 | |
| | | Fish Pot | 4,667 | |
| | | Lobster Pot | 1,382 | |
| | | Other Pot | 2,063 | |
| | 39 | Conch Pots | 1,078,956 | * |
| Striped Bass | | | 12,537 | (July-Sept) |
| Black Sea Bass | 35 | Pots | 625,902 | * |
| | | | | |
| Atlantic Mackerel | | Weir | 430,785 | |
| Squid | | Weir | 322,608 | |
| King Mackerel | | Weir | 151,615 | |
| Scup | | Weir | 76,693 | |
| Butterfish | | Weir | 12,464 | |
| Bluefish | | Weir | 11,076 | |
| Spanish Mackerel | | Weir | 11,046 | |
| Fluke | | Weir | 3,924 | |
| Albacore | | Weir | 1,363 | |
| Bonito | | Weir | 356 | |
| Tautog | | Weir | 51 | |
| Amberjack | | Weir | 27 | |
| Weakfish | | Weir | 18 | |
| | | | | |
| Bay scallop (without shell) | | | 17,813 | |
| Bay scallop (with shell) | | | 28,068 | |
| Littlenecks | | | 200 | |
| Mixed quahogs | | | 3,985 | |
| Mussels | | | 8,548,273 | |
| Sea clams | | | 12,816,980 | |
| Soft shell clams | | | 42,285 | |
| Sea scallops (without shell) | | | 413 | |

* From state catch report data

Estimates of the numbers of recreational angler trips are in the hundreds of thousands, according to the Massachusetts Division of Marine Fisheries.

## *Season*

Spring through fall is when most commercial fishing takes place. "May to December there's someone making money all the time . . . . There is not much activity on the Sound in the winter."

## *Bottom Type*

The vast majority (90%) of the Sound is sandy and the sand ridges that form seem to "hold a lot of fish." There are pockets of mud, usually in the deepest portions of the Sound.

## *Impacts of the Proposed Wind Farm On:*

### Fishing

The consensus of those interviewed for this project is that if the wind farm is built as planned, it will close the most productive portion of the Sound's fishing grounds to the mobile gear fishing fleet. The footprint would take up about 1/3 of the active vessels' fishing grounds, but could diminish their landings by two-thirds (see charts in Appendix 2). Asked if why they couldn't simply fish in the areas that would remain open, the fishermen emphatically said, "That is not where the fish are!" Also, Massachusetts' three-mile restriction on dragging already eliminates some areas for mobile gear.

The design of the wind farm precludes the possibility of towing between the towers. Gear is towed at distances up to 1000' or more behind the boat. "It is impractical to think that you know exactly where the head is, considering the way the tide goes, the way the boat goes . . . ." The tide and currents are strong and constantly moving the gear, sometimes in unanticipated ways. Moreover, the grid pattern of the farm could lead to dangerous gear conflicts between mobile fishermen according to participants.

If fishermen are forced out of this productive area, they may crowd other fishermen. For example, one fisherman noted that he already has to go lobstering in June rather than July because of sea bass closures, so if he is forced out of the Sound, he may have to spend more time lobstering. Other fishermen noted that if the mobile gear fleet is forced off Horseshoe Shoals, they "most certainly will fish closer to shore amongst the smaller hook fleet." Such displacement would increase the potential for gear conflicts. Fishermen have worked out ways to communicate with each other to minimize gear conflicts in the Sound in general and on Horseshoe Shoals in particular. The wind farm construction could create new conflicts by disrupting the traditional fishing patterns.

### Bottom

Some wonder if the structures will cause erosion, given the strength of the currents and the effects of jetties on coastal erosion. It was speculated that this could also affect the ability of the shoals to continue serving as nursery areas for fish. As far as the authors have been able to determine, no formal assessment by a government fisheries agency of this potential impact has been conducted.

### Ownership of Bottom

Permits for any use of the bottom should not confer ownership since it is a common property resource, several fishermen noted. If the project goes forward, some participants suggested that the area could be leased for a set time period at a reasonable market rate. (Author's note: bottom leases are common for shellfish propagation.) Some of the funds thus generated, it was suggested, could be set aside for reevaluation of the impacts and possible compensation to the communities or individuals affected.

### Birds

Concern was expressed particularly for migrating Red Knots (sandpipers), Eider ducks and Roseate terns. The Northeast breeding population of Roseate terns has been listed as "endangered" and 50% of the regional breeding population nests on Bird Island, Massachusetts. Eider ducks are the Northeast's largest duck and they move in great flocks to harvest mussels in the area. It is anticipated that mussels will attach to the windmill platforms and thus will attract Eider ducks that will be likely to be injured or killed by the turning blades. Red Knots are small birds that migrate from the Arctic Circle to South America whose numbers are already far below their historical population. Their tendency to form large concentrations at traditional staging areas during migration makes the population vulnerable.

## *Requests*

Before permitting goes forward, interviewed fishermen urgently requested that in addition to the information provided by this report, that the Commonwealth of Massachusetts consider additional data sets in their analysis, even if this requires additional research. In particular, they requested:

- A review of state records on landings;
- State biologists provide a more accurate assessment of existing stocks; and
- Bottom habitat and sea life be mapped prior to any leasing of the bottom.

## *Conclusions*

The commercial and recreational harvest of fish and shellfish in Nantucket Sound provides millions of dollars in revenue to the local economy and is a way of life in many local towns. One hundred, twenty-three commercial vessels have been identified by name as fishing in Nantucket Sound. Interviewees estimated that mobile gear commercial fishing vessels earn one-half to 60 percent of their annual income fishing on Horseshoe Shoals, the proposed area of the wind farm.

The participants in this project warned that the presence of wind turbines on the Shoals would certainly force some existing commercial fishing businesses to move their activities into other areas of the Sound. In addition, these businesses would probably have to target alternative species, possibly including species that are already fully exploited in the Sound. They believe that potential impacts include gear conflicts, overfishing, and economic losses.

Covering 24 square miles, the Cape Wind project seeks to build one of the world's largest offshore wind power plants on Horseshoe Shoals in Nantucket Sound. The plant will consist of 130 wind turbines connected to a central service platform including a helicopter pad and crew quarters. Each turbine will have about 150 gallons of hydraulic oil and the service platform will have at least 30,000 gallons of dielectric oil and diesel fuel. The plant will be less than 5 miles from land at its closest point. Serious potential environmental impacts identified by participating representatives of the fishing industry included:

- loss of resources due to habitat disruption, pollution
- large-scale habitat conversion of shoals area due to changes in water flow and sediment transport
- increased bird mortality due to strikes and loss of forage
- loss or alteration of critical squid spawning habitat and/or
- loss of fishing access, particularly to mobile gear.

This limited study does not purport to have determined the full scope of the potential impacts of the proposed wind farm on the portion of the fishing industry or fishing communities associated with the use of Horseshoe Shoals. Nor can the authors assert how many individual businesses will be affected, either directly or indirectly. Nevertheless, the authors do caution that a number of mobile gear fishing vessels will be displaced if the proposed Cape Wind farm is constructed, and this displacement could have a broader impact throughout the entire Nantucket Sound area.

As the population of the United States continues to migrate to the coastal zone, conflicting demands for use of both the shoreside and coastal resources will only increase. It is essential that a procedure be put in place to assure that the direct, indirect and cumulative impacts on existing or traditional uses are considered prior to construction of new projects to assure that losses are not incurred, especially when these could be irreversible.

## *Appendix 1: Comment Letter from Captain William Amaru*

Capt. William Amaru
P. O. 1019, 25 Portanimicut Rd.
South Orleans, MA 02662
October 16, 2004

Karen K. Adams
Regulatory Office
New England District, Army Corps of Engineers
696 Virginia Rd.
Concored, MA 01742-2751

Dear Karen:

I am a commercial fisherman from Cape Cod who, together with my son, operate a stern trawler from Chatham. We fish the waters of Nantucket Sound in the Horseshoe Shoals area of the proposed wind energy projecct. I was asked by Wayne Kurker of Hyannis Marine to share my knowledge of the fishery to enable him to provide your office with information about how the wind turbine placement will affect our activities.

A small group of fishermen gathered with Wayne recently and helped to compose a letter with several points concerning the impacts the proposed turbine placement will have. It was clear to me that your office may not have all the information necessary to make a well informed judgement about how trawlers will be affected. The following is a brief description of how our daily operations work.

A trawler tows a series of cables attached to doors which weight and spread the net and keep it on the bottom. The cables are towed behind the boat at a distance of between four and six hundred feet, and the net can be as much as fourteen hundred feet behind the boat. While there is much more to the operation than I can briefly describe, let it be understood that a great deal of space is necessary to safely trawl and maneuver in this fishery. The proposal to place the turbines as close together as described by Wind Associates will place in jeapordy the operators and crews of trawlers. Additionally, boat traffic such as ferries, sail boats, recreational fishers and pleasure boat operators, all of whom share the resource with us, will be placed at greater risk.

I would urge you to refer to Wayne's letter of Oct. 15, 2004 for the technical requirements of this fishery. The Massachusetts Division of Marine Fisheries can supply you with any and all information about the species caught, and their values to the fleet, in the aforementioned area.

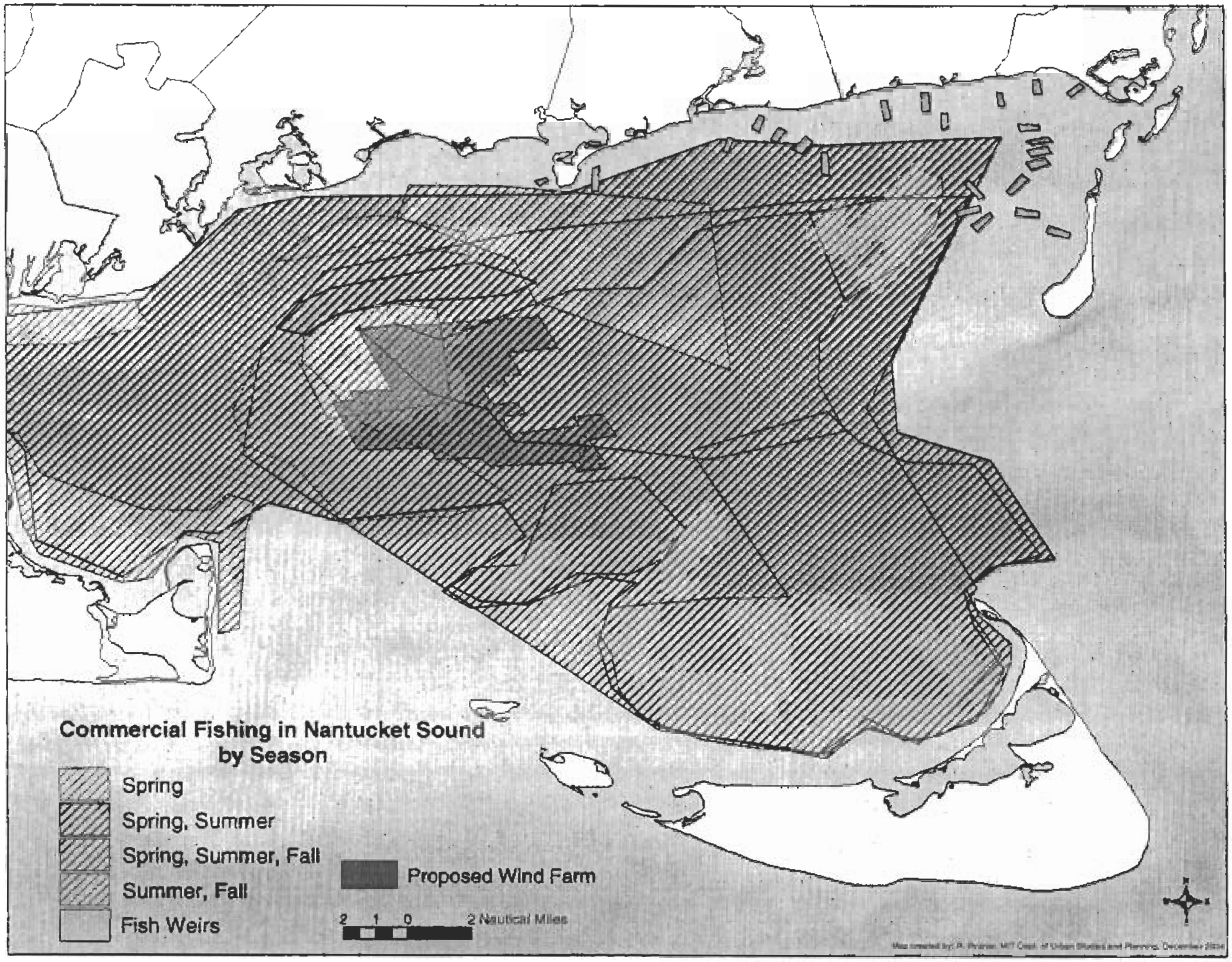
To conclude, despite the lack of information to date, it is imperative that the needs of a significant number of fishers must be taken into consideration when evaluating site use. We as a profession have been asked to give up more than any other user group. The loss of this important fishery would be devastating, and unnecessary. Please contact us if you have any further questions.

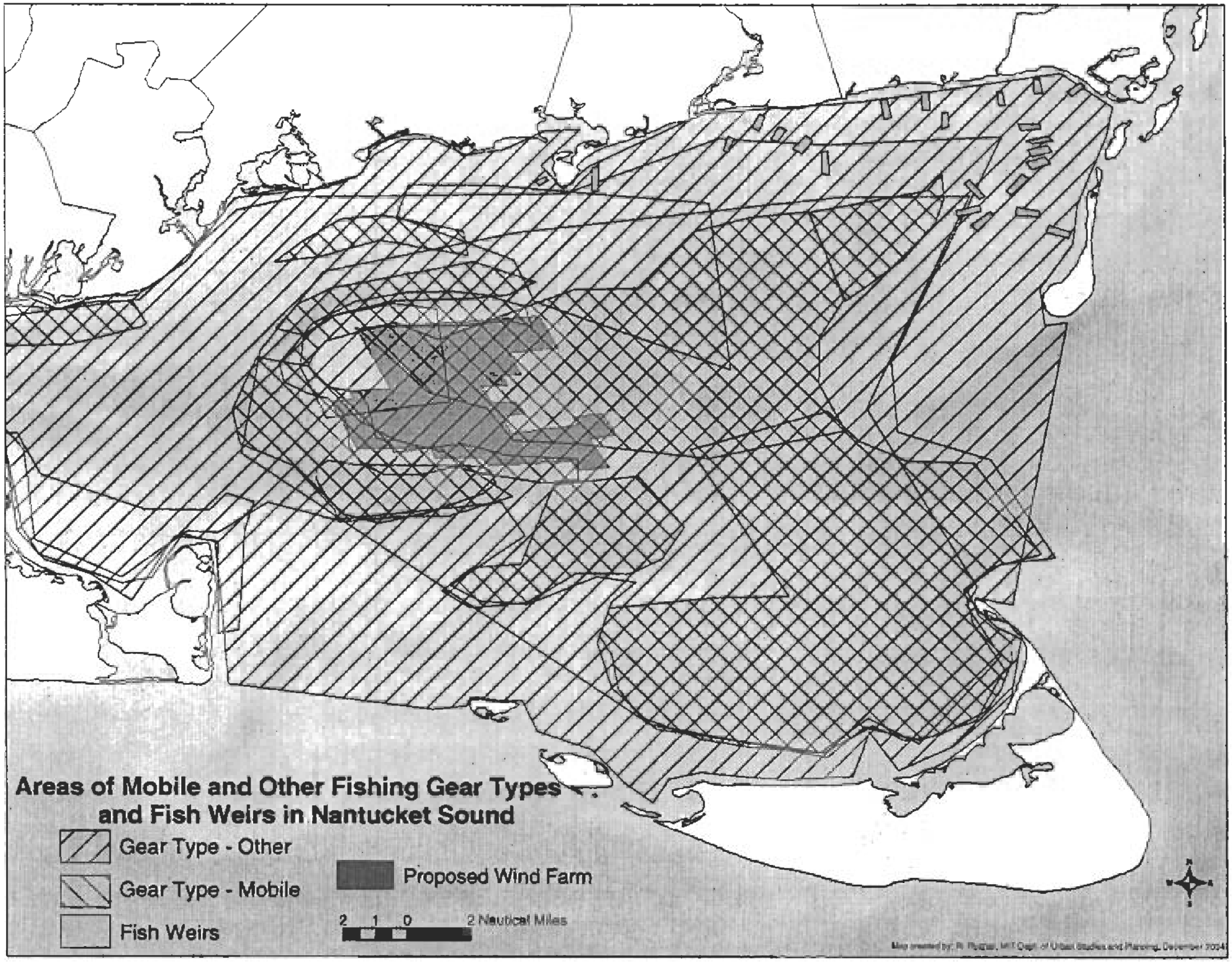
Sincerely, William H. Amaru

William H. Amaru

cc: Alliance to Protect Nantucket Sound

## *Appendix 2: Aggregate charts of commercial fishing in Nantucket Sound*


Commercial Fishing in Nantucket Sound
by Season
Spring
Spring, Summer
Spring, Summer, Fall
Summer, Fall
Fish Weirs
Proposed Wind Farm
2 1 0 2 Nautical Miles


Areas of Mobile and Other Fishing Gear Types
and Fish Weirs in Nantucket Sound
Gear Type - Other
Gear Type - Mobile
Fish Weirs
Proposed Wind Farm
2 1 0 2 Nautical Miles

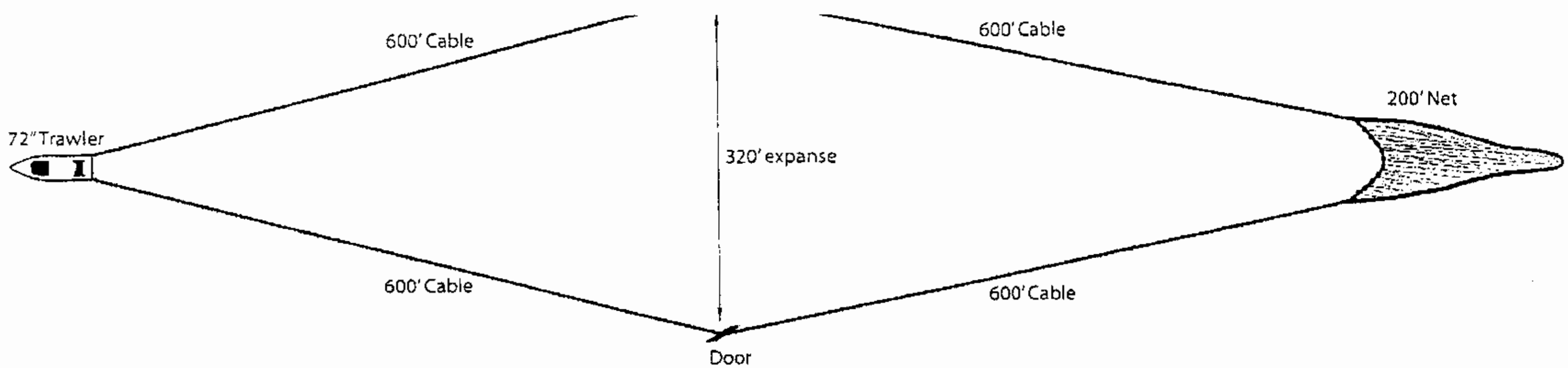
10/15/2004

## 40' COMMERCIAL FISHING TRAWLER

| | SQUID<br>BUTTER FISH | FLUKE SEABASS<br>SCUP BLUEFISH<br>CONCH<br>HORSEHOE CRAB |
|---|---|---|
| TOW LINE | 420' + | 420' + |
| GROUND CABLE | 180' + | 420' + |
| NET | 175' + | 100' + |
| | 775' | 940' |
| | APPROX. 800' | APPROX.1000' |

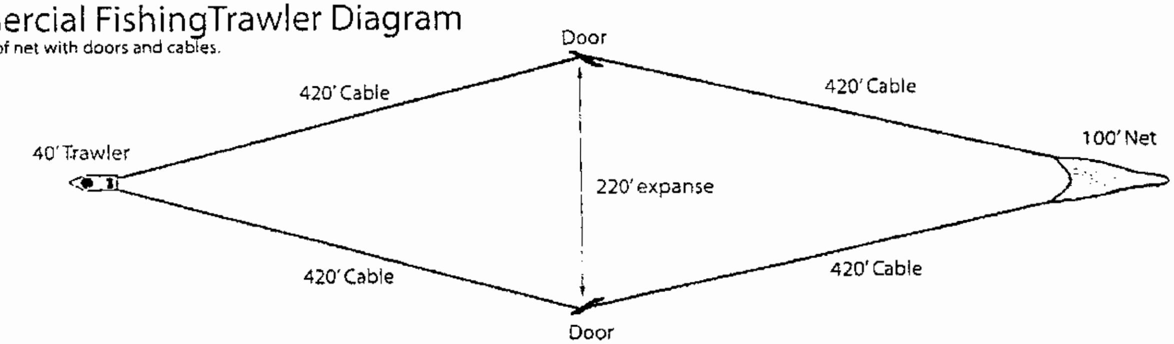
## 72' COMMERCIAL FISHING TRAWLER

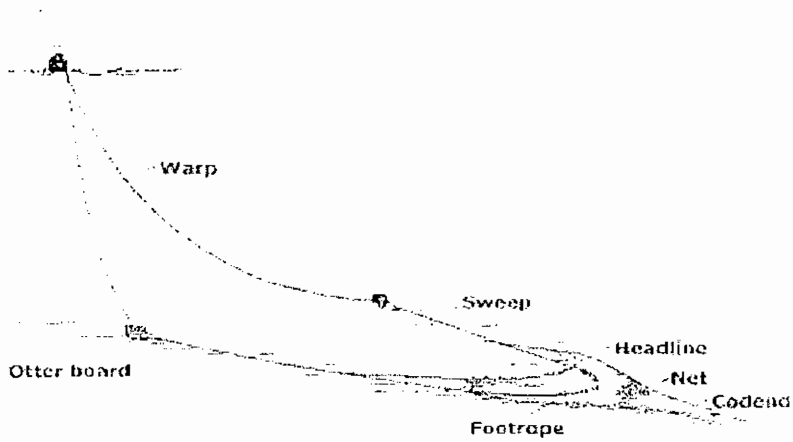
| | SQUID<br>BUTTER FISH | FLUKE SEABASS<br>SCUP BLUEFISH<br>CONCH<br>HORSEHOE CRAB |
|---|---|---|
| TOW LINE | 600' + | 600' + |
| GROUND CABLE | 400' + | 600' + |
| NET | 200' + | 200' + |
| | APPROX 1,200' | APPROX.1,400' |




# 40' Commercial FishingTrawler Diagram

showing full extent of net with doors and cables.




Warp
Sweep
Headline
Otter board
Net
Codend
Footrope

## *Appendix 3: Chatham's Tally*

According to a tally by a fishing business in Chatham, fifty-four commercial fishing vessels from Chatham could be affected by the construction of the wind farm in Nantucket Sound. In addition, the same business counted twelve vessels from Harwich, six from Orleans, five each from Nantucket and Marshfield, four each from Gloucester and Brewster, three each from Edgartown (Martha's Vineyard), and Falmouth, two each from Rockport, Barnstable, Yarmouth and Newport, RI and one each from 18 other communities that might be affected. In addition to the crews of the fishing vessels tallied, another 72 individuals, including shellfishermen, and thirteen fishing organizations could be affected. Moreover, there are four fish weir companies that are currently operating in Nantucket Sound with weir grants in the towns of Chatham, Harwich, Dennis, Yarmouth, Hyannis, Centerville, and Osterville.

**Number of vessels that fish in Nantucket Sound listed by community**

| Total Number of Vessels | | | | | | | |
|---|---|---|---|---|---|---|---|
| One | Two | Three | Four | Five | Six | Twelve | Fifty-four |
| Auburn | Rockport | Edgartown | Gloucester | Nantucket | Orleans (including S. Orleans) | Harwich (including S.Harwich, and Harwichport | Chatham (including W. Chatham, N. Chatham, S. Chatham) |
| Holden | W. Barnstable | Falmouth | Brewster | Marshfield | | | |
| Norwood | W. Yarmouth | | | | | | |
| Stoneham | Newport, RI | | | | | | |
| Duxbury | | | | | | | |
| Hanover | | | | | | | |
| Wareham | | | | | | | |
| Cotuit | | | | | | | |
| E. Orleans | | | | | | | |
| Eastham | | | | | | | |
| Osterville | | | | | | | |
| S. Dennis | | | | | | | |
| Wellfleet | | | | | | | |
| Assonet | | | | | | | |
| Mattapoisett | | | | | | | |
| Raynham | | | | | | | |
| Rochester | | | | | | | |
| Rockaway, NY | | | | | | | |
| Gramby | | | | | | | |

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# Valuing Ecosystem Services

## Toward Better Environmental Decision-Making

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NOT FOR PUBLIC RELEASE BEFORE

***Tuesday, October 26, 2004***

9:00 a.m. EDT

This prepublication version of the report has been provided to the public to facilitate timely access to the committee's conclusions and recommendations. Although the substance of the report is final, editorial changes may be made prior to publication. The final report will be available through the National Academies Press later this year.

NATIONAL RESEARCH COUNCIL

*OF THE NATIONAL ACADEMIES*

PREPUBLICATION COPY

# Valuing Ecosystem Services

## Toward Better Environmental Decision-Making

Committee on Assessing and Valuing the Services of Aquatic and Related Terrestrial Ecosystems

Water Science and Technology Board

Division on Earth and Life Studies

NATIONAL RESEARCH COUNCIL
OF THE NATIONAL ACADEMIES

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## COMMITTEE ON ASSESSING AND VALUING THE SERVICES OF AQUATIC AND RELATED TERRESTRIAL ECOSYSTEMS

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# Preface

The development of the ecosystem services paradigm has enhanced our understanding of how the natural environment matters to human societies. We now think of the natural environment, and the ecosystems of which it consists, as natural capital—a form of capital asset that, along with physical, human, social, and intellectual capital, is one of society's important assets. As President Theodore Roosevelt presciently said in 1907,

> The nation behaves well if it treats the natural resources as assets which it must turn over to the next generation increased and not impaired in value.[1]

Economists normally value assets by the value of services that they provide: Can we apply this approach to ecological assets by valuing the services provided by ecosystems?

An ecosystem is generally accepted to be an interacting system of biota and its associated physical environment. Aquatic and related terrestrial ecosystems are among the most important ecosystems in the United States, and Congress through the Clean Water Act has recognized the importance of the services they provide and has shown a concern that these services be restored and maintained. Such systems intuitively include streams, rivers, ponds, lakes, estuaries, and oceans. However, most ecologists and environmental regulators include vegetated wetlands as aquatic ecosystems, and many also think of underlying groundwater aquifers as potential members of the set. Thus, the inclusion of "related terrestrial ecosystems" for consideration in this study is a reflection of the state of the science that recognizes the multitude of processes linking terrestrial and aquatic systems.

Many of the policies implemented by various federal, state, and local regulatory agencies can profoundly affect the nation's aquatic and related terrestrial ecosystems, and in consequence, these bodies have an interest in better understanding the nature of their services, how their own actions may affect them, and what value society places on their services. The need for this study was recognized in 1997 at a strategic planning session of Water Science and Technology Board (WSTB) of the National Research Council (NRC). The Committee on the Assessing and Valuing the Services of Aquatic and Related Terrestrial Ecosystems was established by the NRC in early 2002 with support from the U.S. Environmental Protection Agency (EPA), U.S. Army Corps of Engineers (USACE), and U.S. Department of Agriculture (USDA). Its members are drawn from the ranks of economists, ecologists, and philosophers who have professional expertise relating to aquatic ecosystems and the valuation of ecosystem services.

In drafting this report the committee members have sought to understand and integrate the disciplines, primarily ecology and economics, that cover the field of ecosystem service valuation. In fact, the committee quickly discovered that this is not an established field—ecologists have only recently begun to think in terms of ecosystem services and their determinants, while economists have likewise only very recently begun to incorporate the factors

[1] Quoted on the wall of the entrance hall of the American Museum of Natural History.

affecting ecosystem services into their valuations of these services. If we as a society are to understand properly the value of our natural capital, which is a prerequisite for sensible conservation decisions, then this growing field must be developed further and this report provides detailed recommendations for facilitating that development. Although the field is relatively new, a great deal is understood, and consequently the committee makes many positive conclusions and recommendations concerning the methods that can be applied in valuing the services of aquatic and related terrestrial ecosystems. Furthermore, because the principles and practices of valuing ecosystem services are rarely sensitive to whether the underlying ecosystem is aquatic or terrestrial, the report's various conclusions and recommendations are likely to be directly, or at least indirectly applicable to valuation of the goods and services provided by any ecosystem.

The study benefited greatly from the knowledge and expertise of those who made presentations at our meetings, including Richard Carson, University of California, San Diego; Harry Kitch, USACE; John McShane, EPA; Angela Nugent, EPA; Michael O'Neill, USDA; Mahesh Podar, EPA (retired); John Powers, EPA; Stephen Schneider, Stanford University; and Eugene Stakhiv, USACE Institute for Water Resources. The success of the report also depended on the support of the NRC staff working with the committee, and it is a particular pleasure to acknowledge the immense assistance of Study Director Mark Gibson and WSTB Research Associate Ellen de Guzman. Finally, of course, the committee members worked extraordinarily hard and with great dedication, expertise, and good humor in pulling together what was initially a rather disparate set of issues and methods into the coherent whole that follows.

This report was reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise in accordance with the procedures approved by the NRC's Report Review Committee. The purpose of this independent review is to provide candid and critical comments that will assist the institution in making its published report as sound as possible and to ensure that the report meets institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the deliberative process. We wish to thank the following individuals for their review of this report: Mark Brinson, East Carolina University, Greenville, North Carolina; J. Baird Callicott, University of North Texas, Denton; Nancy Grimm, Arizona State University, Tempe; Michael Hanemann, University of California, Berkeley; Peter Kareiva, The Nature Conservancy, Seattle, Washington; Raymond Knopp, Resources for the Future, Washington, D.C.; Sandra Postel, Global Water Policy Project, Amherst, Massachusetts; and Robert Stavins, Harvard University, Cambridge.

Although the reviewers listed above have provided many constructive comments and suggestions, they were not asked to endorse the conclusions or recommendations, nor did they see the final draft of the report before its release. The review of this report was overseen by John Boland, Johns Hopkins University, Baltimore. Appointed by the National Research Council, he was responsible for making certain that an independent examination of the report was carefully carried out in accordance with institutional procedures and that all review comments were carefully considered. Responsibility for the final content of this report rests entirely with the authoring committee and the NRC.

Geoffrey M. Heal, *Chair*

# Contents

# Executive Summary

## OVERVIEW

Ecosystems provide a wide variety of marketable goods, fish and lumber being two familiar examples. However, society is increasingly recognizing the myriad functions—the observable manifestations of ecosystem processes such as nutrient recycling, regulation of climate, and maintenance of biodiversity—that they provide, without which human civilizations could not thrive. Derived from the physical, biological, and chemical processes at work in natural ecosystems, these functions are seldom experienced directly by users of the resource. Rather, it is the services provided by ecosystems, such as flood risk reduction and water supply, together with ecosystem goods, that create value for human users and are the subject of this report.[1]

Aquatic ecosystems include freshwater, marine, and estuarine surface waterbodies. These incorporate lakes, rivers, streams, coastal waters, estuaries, and wetlands, together with their associated flora and fauna. Each of these entities is connected to a greater ecological and hydrological landscape that includes adjacent riparian areas, upland terrestrial ecosystems, and underlying groundwater aquifers. Thus, the term "aquatic ecosystems" in this report includes these related terrestrial ecosystems and underlying aquifers. Aquatic ecosystems perform numerous interrelated environmental functions and provide a wide range of important goods and services. Many aquatic ecosystems enhance the economic livelihood of local communities by supporting commercial fishing and agriculture and by serving the recreational sector. The continuance or growth of these types of economic activities is directly related to the extent and health of these natural ecosystems.

However, human activities, rapid population growth, and industrial, commercial, and residential development have all led to increased pollution, adverse modification, and destruction of remaining (especially pristine) aquatic ecosystems—despite an increase in federal, state, and local regulations intended to protect, conserve, and restore these natural resources. Increased human demand for water has simultaneously reduced the amount available to support these ecosystems. Notwithstanding the large losses and changes in these systems, aquatic ecosystems remain broadly and heterogeneously distributed across the nation. For example, there are almost 4 million miles of rivers and streams, 59,000 miles of ocean shoreline waters, and 5,500 miles of Great Lakes shoreline in the United States; there are 87,000 square miles of estuaries, while lakes, reservoirs, and ponds account for more than 40 million acres.

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[1] *Ecosystem structure* refers to both the composition of the ecosystem (i.e., its various parts) and the physical and biological organization defining how those parts are organized. A leopard frog or a marsh plant such as a cattail, for example, would be considered a component of an aquatic ecosystem and hence part of its structure. *Ecosystem function* describes a process that takes place in an ecosystem as a result of the interactions of the plants, animals, and other organisms in the ecosystem with each other or their environment. Primary production (the process of converting inorganic compounds into organic compounds by plants, algae, and chemoautotrophs) is an example of an ecosystem function. Ecosystem structure and function provide various *ecosystem goods* and *services* of value to humans such as fish for recreational or commercial use, clean water to swim in or drink, and various esthetic qualities (e.g., pristine mountain streams or wilderness areas) (see Box 3-1 for further information).

Despite growing recognition of the importance of ecosystem functions and services, they are often taken for granted and overlooked in environmental decision-making. Thus, choices between the conservation and restoration of some ecosystems and the continuation and expansion of human activities in others have to be made with an enhanced recognition of this potential for conflict and of the value of ecosystem services. In making these choices, the economic values of the ecosystem goods and services must be known so that they can be compared with the economic values of activities that may compromise them and so that improvements to one ecosystem can be compared to those in another.

This report was prepared by the National Research Council (NRC) Committee on Assessing and Valuing the Services of Aquatic and Related Terrestrial Ecosystems, overseen by the NRC's Water Science and Technology Board, and supported by the U.S. Army Corps of Engineers, U.S. Environmental Protection Agency, and the U.S. Department of Agriculture (see Box ES-1). The committee consisted of 11 volunteer experts drawn from the fields of ecology, economics, and philosophy who have professional expertise relating to aquatic ecosystems and to the valuation of ecosystem services. This report's contents, conclusions, and recommendations are based on a review of relevant technical literature, information gathered at five committee meetings, and the collective expertise of committee members. Because of space limitations, this Executive Summary includes only the major conclusions and related recommendations of the committee in the general order of their appearance in the report. More detailed conclusions and recommendations can be found throughout the report.

Valuing ecosystem service requires the successful integration of ecology and economics and presents several challenges that are discussed throughout this report. The fundamental

**BOX ES-1**
**Statement of Task**

The committee will evaluate methods for assessing services and the associated economic values of aquatic and related terrestrial ecosystems. The committee's work will focus on identifying and assessing existing economic methods to quantitatively determine the intrinsic value of these ecosystems in support of improved environmental decision-making, including situations where ecosystem services can be only partially valued. The committee will also address several key questions, including:

- What is the relationship between ecosystem services and the more widely studied ecosystem functions?
- For a broad array of ecosystem types, what services can be defined, how can they be measured, and is the knowledge of these services sufficient to support an assessment of their value to society?
- What lessons can be learned from a comparative review of past attempts to value ecosystem services—particularly, are there significant differences between eastern and western U.S. perspectives on these issues?
- What kinds of research or syntheses would most rapidly advance the ability of natural resource managers and decision makers to recognize, measure, and value ecosystem services?
- Considering existing limitations, error, and bias in the understanding and measurement of ecosystem values, how can available information best be used to improve the quality of natural resource planning, management, and regulation?

challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structures and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values (see Figure ES-1).

Ecosystems are complex however, making the translation from ecosystem structure and function to ecosystem goods and services (i.e., the ecological production function) difficult. Similarly, in many cases the lack of markets and market prices and of other direct behavioral links to underlying values makes the translation from quantities of goods and services to value (and the direct translation from ecosystem structure to value) quite difficult, though both are given by an economic valuation function. Probably the greatest challenge for successful valuation of ecosystem services is to integrate studies of the ecological production function with studies of the economic valuation function. To do this, the definitions of ecosystem goods and services must match across studies. Failure to do so means that the results of ecological studies cannot be carried over into economic valuation studies. Attempts to value ecosystem services without this key link will either fail to have ecological underpinnings or fail to be relevant as valuation studies.

Where an ecosystem's services and goods can be identified and measured, it will often be possible to assign values to them by employing existing economic valuation methods. The emerging desire to measure the environmental costs of human activities, or to assess the benefits of environmental protection and restoration, has challenged the state of the art in environmental evaluation in both the ecological and the social sciences. Some ecosystem goods and services cannot be valued because they are not quantifiable or because available methods are not appropriate or reliable. Economic valuation methods can be complex and demanding, and the results of applying these methods may be subject to judgment, uncertainty, and bias. However, based on an assessment of a very large literature on the development and application of various economic valuation methods, the committee concludes that they are mature and capable of providing useful information in support of improved environmental decision-making.

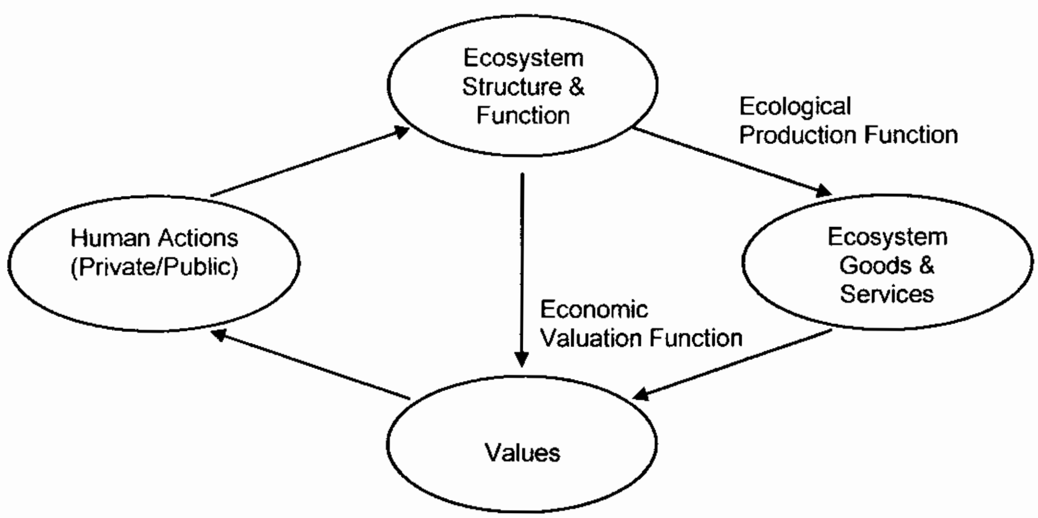


FIGURE ES-1 Components of ecosystem valuation: ecosystem structure and function, goods and services, human actions, and values. (See Figure 7-1 for an expanded version of this figure.)

From an ecological perspective, the challenge is to interpret basic research on ecosystem functions so that service-level information can be communicated to economists. For economic and related social sciences, the challenge is to identify the values of both tangible and intangible goods and services associated with ecosystems and to address the problem of decision-making in the presence of partial valuation. The combined challenge is to develop and apply methods to assess the values of human-induced changes in aquatic and related terrestrial ecosystem functions and services.

Finally, this report concerns valuing the goods and services that ecosystems provide to human societies, with principal focus on those provided by aquatic and related terrestrial ecosystems. However, because the principles and practices of valuing ecosystem goods and services are rarely sensitive to whether the underlying ecosystem is strictly aquatic or terrestrial, many of the report's conclusions and recommendations are likely to be directly or at least indirectly applicable to the valuation of goods and services provided by any ecosystem.

## THE MEANING OF VALUE AND USE OF ECONOMIC VALUATION IN THE ENVIRONMENTAL POLICY DECISION-MAKING PROCESS

In order to develop a perspective on valuing aquatic ecosystems, it is necessary to first provide a clear discussion and statement of what it means to value something and of the role of "valuation" in environmental policymaking. In this regard, environmental issues and ecosystems have been at the core of many recent philosophical discussions regarding value (see Chapter 2). Fundamentally, these debates about the value of ecosystems derive from two points of view. The first is that the values of ecosystems and their services are non-anthropocentric and that nonhuman species have moral interests or rights unto themselves. The other, which includes the economic approach to valuation, is that all values are anthropocentric. This report focuses on the sources of value that can be captured through economic valuation.[2] However, the committee recognizes that all forms of value may ultimately contribute to decisions regarding ecosystem use, preservation, or restoration.

Although economic valuation does not capture all sources or types of value (e.g., intrinsic values on which the notion of rights is founded), it is much broader than usually presumed. It recognizes that economic value can stem from the use of an environmental resource (use values), including both commercial and noncommercial uses, or from its existence even in the absence of use (nonuse value). The broad array of values included under this approach is captured by using the total economic value (TEV) framework to identify potential sources of this value. Use of the TEV framework helps to provide a checklist of potential impacts and effects that need to be considered in valuing ecosystem services as comprehensively as possible. By its nature, economic valuation involves the quantification of values based on a common metric, normally a monetary metric. The use of a dollar metric for quantifying values is based on the assumption that individuals are willing to trade the ecological service being valued for more of other goods and services represented by the metric (more dollars). Use of a monetary metric allows measurement of the costs or benefits associated with changes in ecosystem services.

The role of economic valuation in environmental decision-making depends on the specific criteria used to choose among policy alternatives. If policy choices are based primarily

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[2] Unless otherwise noted, use of the terms "value," "valuing," or "valuation" refers to economic valuation, more specifically, the economic valuation of ecosystem goods and services.

on intrinsic values, there is little need for the quantification of values through economic valuation. However, if policymakers consider trade-offs and benefits and costs when making policy decisions, then quantification of the value of ecosystem services is essential. Failure to include some measure of the value of ecosystem services in benefit-cost calculations will implicitly assign them a value of zero. The committee believes that considering the best available and most reliable information about the benefits of improvements in ecosystem services or the costs of ecosystem degradation will lead to improved environmental decision-making. The committee recognizes, however, that this information is likely to be only one of many possible considerations that influence policy choice.

The benefit and cost estimates that emerge from an economic valuation exercise will be influenced by the way in which the valuation question is framed. In particular, the estimates will depend on the delineation of changes in ecosystem goods or services to be valued, the scope of the analysis (in terms of both the geographical boundaries and the inclusion of relevant stakeholders), and the temporal scale. In addition, the valuation question can be framed in terms of two alternative measures of value, willingness to pay (WTP) and willingness to accept (compensation) (WTA). These two approaches imply different presumptions about the distribution of property rights and can differ substantially, depending on the availability of substitutes and income limitations. In many contexts, methodological limitations necessitate the use of WTP rather than WTA.

Finally, because ecosystem changes are likely to have long-term impacts, some accounting of the timing of impacts is necessary. This can be done through discounting future costs and benefits. It is essential, however, to recognize that consumption discounting is distinct from the discounting of utility, which reflects the weights put on the well-being of different generations.

Based on these conclusions, the committee makes the following recommendations (Chapter 2):

- Policymakers should use economic valuation as a means of evaluating the trade-offs involved in environmental policy choices; that is, an assessment of benefits and costs should be part of the information set available to policymakers in choosing among alternatives.
- If the benefits and costs of a policy are evaluated, the benefits and costs associated with changes in ecosystem services should be included along with other impacts to ensure that ecosystem effects are adequately considered in policy evaluation.
- Economic valuation of changes in ecosystem services should be based on the comprehensive definition embodied in the TEV framework; both use and nonuse values should be included.
- The valuation exercise should be framed properly. In particular, it should value the *changes* in ecosystem good or services attributable to a policy change.
- In the aggregation of benefits and/or costs over time, the consumption discount rate, reflecting changes in scarcity over time, should be used instead of the utility discount rate.

## AQUATIC AND RELATED TERRESTRIAL ECOSYSTEMS

An ecosystem is generally accepted to be an interacting system of biota and its associated physical environment; ecologists tend to think of these systems as identifiable at many different scales with boundaries selected to highlight internal and external interactions. The phrase "aquatic and related terrestrial ecosystems" recognizes the impossibility of analyzing aquatic systems absent consideration of the linkages to adjacent terrestrial environments. For many of the ecosystem functions and derived services considered in this report, it is not possible, necessary, or appropriate to delineate clear spatial boundaries between aquatic and related terrestrial systems (see also Box 3-1). Indeed, to the extent there is an identifiable boundary, it is often dynamic in both space and time.

The conceptual challenges of valuing ecosystem services are explicit description and adequate assessment of the link between the structure and function of natural systems and the goods or services derived by humanity (see Figure ES-1). Describing structure is a relatively straightforward process, even in highly diverse ecosystems. However, ecosystem functions are often difficult to infer from observed structure in natural systems. Furthermore, the relationship between structure and function, as well as how these attributes respond to disturbance, are not often well understood. Without comprehensive understanding of the behavior of aquatic systems, it is clearly difficult to describe thoroughly all of the services these systems provide society. Although valuing ecosystem services that are not completely understood is possible (see more below), when valuation becomes an important input in environmental decision-making, there is the risk that it may be incomplete.

There have only been a few attempts to develop explicit maps of the linkage between aquatic ecosystem structure/function and value. There are, however, a multitude of efforts to separately identify ecosystem functions, goods, services, values, and/or other elements in the linkage, without developing a comprehensive argument. One consequence of this disconnect is a diverse literature that suffers somewhat from indistinct terminology, highly variable perspectives, and considerable, divergent convictions. However, the development of an interdisciplinary terminology and a universally applicable protocol for valuing aquatic ecosystems was ultimately identified by the committee as unnecessary. From an ecological perspective, the value of specific ecosystem functions/services is entirely relative. The spatial and temporal scales of analysis are critical determinants of potential value. Ecologists have described the structure and function of most types of aquatic ecosystems qualitatively, and general concepts regarding the linkages between ecosystem function and services have been developed. Although precise quantification of these relationships remains elusive, the general concepts seem to offer sufficient guidance for valuation to proceed with careful attention to the limitations of any ecosystem assessment. Further integration of economics and ecology at both intellectual and practical scales will improve ecologists' ability to provide useful information for assessing and valuing aquatic ecosystems.

There remains a need for a significant amount of research in the ongoing effort to codify the linkage between ecosystem structure and function and the provision of goods and services for subsequent valuation. The complexity, variability, and dynamic nature of aquatic ecosystems make it likely that a comprehensive identification of all functions and derived services may never be achieved. Nevertheless, comprehensive information is not generally necessary to inform management decisions. Despite this unresolved state, future ecosystem valuation efforts can be improved through use of several general guidelines and by research in the following areas (Chapter 3):

- Aquatic ecosystems generally have some capacity to provide consumable resources, habitat for plants and animals, regulation of the environment, and support for nonconsumptive uses, and considerable work remains to be done in documentation of the potential of various aquatic ecosystems for contribution in each of these broad areas.

- Because delivery of ecosystem goods and services occurs in both space and time, investigation of the spatial and temporal thresholds of significance for various ecosystem services is necessary to inform valuation efforts.
- Natural systems are dynamic and frequently exhibit nonlinear behavior, and caution should be used in extrapolation of measurements in both space and time. Although it is not possible to avoid all mistakes in extrapolation, the uncertainty warrants explicit acknowledgment. Methods are needed to assess and articulate this uncertainty as part of system valuations.

## METHODS OF NONMARKET VALUATION

In response to the committee's statement of task (see Box ES-1), this report outlines the major nonmarket methods currently available for estimating monetary values of aquatic and related terrestrial ecosystem services. This includes a review of the economic approach to valuation, which is based on the aforementioned TEV framework. In addition to presenting valuation approaches, the applicability of each method to valuing ecosystem services is discussed. All of this is provided within the context of the committees' implicit objective of assessing the literature in order to facilitate original studies that will develop a closer link between aquatic ecosystem functions, services, and value estimates. It is important to note however, that the report does not provide instructions on how to apply each of the methods, but rather provides a rich listing of references that can be used to develop a greater understanding of any of the methods.

There is a variety of nonmarket valuation approaches that are currently available to be applied in valuing aquatic and related terrestrial ecosystem services. Revealed-preference methods (e.g., averting behavior, travel cost, hedonics) can be applied only to a limited number of ecosystem services. However, both the range and the number of services that can potentially be valued are increasing with the development of new methods, such as dynamic production function approaches, general equilibrium modeling of integrated ecological-economic systems, and combined revealed- and stated-preference approaches.

Stated-preference methods, including contingent valuation and conjoint analysis, can be more widely applied, and certain values can be estimated only through the application of such techniques. On the other hand, the credibility of estimated values for ecosystem services derived from stated-preference methods has often been criticized. For example, contingent valuation methods have come under such scrutiny that it led to National Oceanic and Atmospheric Administration guidelines of "good practice" for these methods in the early 1990s.

Benefit transfers and replacement cost and cost of treatment methods are increasingly being used in environmental valuation, although their application to aquatic ecosystem services is still limited. Economists generally consider benefit transfers as to be a "second-best" valuation method and have devised guidelines governing their use. In contrast, replacement cost and cost of treatment methods should be used with great caution if at all. Although economists have attempted to design strict guidelines for using replacement cost as a last resort "proxy" valuation estimation for an ecological service, in practice estimates employing the replacement cost or cost of treatment approach rarely conform to the conditions outlined by such guidelines.

At least three basic questions arise for any method that is chosen to value aquatic ecosystem services. First, are the services that have been valued those that are the most important for supporting environmental decision-making and policy analyses involving benefit-cost analysis, regulatory impact analysis, legal judgments, and so on? Second, can the services of the aquatic ecosystem that are valued be linked in some substantial way to changes in the functioning of the system? Last, are important services provided by aquatic ecosystems that have not yet been valued so that they are not being given full consideration in policy decisions that affect the quantity and quality of these systems? In many ways, the answers to these questions are the most important criteria for judging the overall validity of the valuation method chosen.

Only a limited number of ecosystem services have been valued to date, and effective treatment of aquatic ecosystem services in benefit-cost analyses requires that more services be valued. Nonuse values require special consideration; these may be the largest component of total economic value for aquatic ecosystem services. Unfortunately, nonuse values can be estimated only with stated-preference methods, and this is the application in which these methods have been soundly criticized.

Although a variety of valuation methods are currently available, no single method can be considered best at all times and for all types of aquatic ecosystem applications. In each application it is necessary to consider what method(s) is the most appropriate. Based on its assessment of the current literature and the preceding conclusions, the committee makes the following recommendations (Chapter 4):

- Specific attention should be given to funding research at the "cutting edge" of the valuation field, such as dynamic production function approaches, general equilibrium modeling of integrated ecological-economic systems, conjoint analysis, and combined stated-preference and revealed-preference methods.
- Specific attention should be given to funding research on improved valuation study designs and validity tests for stated-preference methods applied to determine the nonuse values associated with aquatic and related terrestrial ecosystem services.
- Benefit transfers should be considered a "second-best" method of ecosystem services valuation and should be used with caution and only if appropriate guidelines are followed.
- The replacement cost method and estimates of the cost of treatment are not valid approaches to determining benefits and should not be employed to value aquatic ecosystem services. In the absence of any information on benefits, and under strict guidelines, treatment costs could help determine cost-effective policy action.

## TRANSLATING ECOSYSTEM FUNCTIONS TO THE VALUE OF ECOSYSTEM SERVICES: CASE STUDIES AND LESSONS LEARNED

Although there has been great progress in ecology in understanding ecosystem processes and functions, and in economics in developing and applying nonmarket valuation techniques for their subsequent valuation, at present there often remains a gap between the two. There has been mutual recognition among at least some ecologists and economists that addressing issues such as conserving ecosystems and biodiversity requires the input of both disciplines to be successful. Yet there are few examples of studies that have successfully translated knowledge of ecosystems into a form in which economic valuation can be applied in a meaningful way. Several factors contribute to this ongoing lack of integration. First, ecology and economics are separate disciplines—one in the natural sciences, the other in the social sciences. Traditionally, academic organization and the reward structures for scientists make collaboration across disciplinary boundaries difficult even when the desire to do so exists. Second, the concept of ecosystem services and attempts to value them are still relatively recent; building the necessary working relationships and integrating methods across disciplines will take time.

Nevertheless, some useful integrated studies on the value of aquatic and related terrestrial ecosystem goods and services are starting to emerge. Chapter 5 of this report provides a series of case studies of the integration of ecology and economics necessary for valuing the services of aquatic and related terrestrial ecosystems (including those from both the eastern and the western United States; see Box ES-1). More specifically, this review begins with situations in which the focus is on valuing a single ecosystem service. Typically these are cases in which the service is well defined, there is reasonably good ecological understanding of how the service is produced, and there is reasonably good economic understanding of how to value it. Even when valuing a single ecosystem service however, there can be significant uncertainty either about the production of the ecosystem service, the value of the ecosystem service, or both. Next, attempts to value multiple ecosystem services are reviewed. Since ecosystems produce a range of services, and these services are frequently closely connected, it is often hard to discuss valuation of a single service in isolation. However, valuing multiple ecosystem services typically multiplies the difficulty of evaluation. Last to be reviewed are analyses that attempt to encompass all services produced by an ecosystem. Such cases can arise with natural resource damage assessment, where a dollar value estimate of total damages is required, or with ecosystem restoration efforts, and will typically face large gaps in understanding and information in both ecology and economics.

Proceeding from single services to entire ecosystems illustrates the range of circumstances and methods for valuing ecosystem goods and services. In some cases, it may be possible to generate relatively precise estimates of value. In other cases, all that may be possible is a rough categorization (e.g., "a lot" versus "a little"). Whether there is sufficient information for the valuation of ecosystem services to be of use in environmental decision-making depends on the circumstances and the policy question or decision at hand (see Chapters 2 and 6 for further information). In a few instances, a rough estimate may be sufficient to decide that one option is preferable to another. Tougher decisions will typically require more refined understanding of the issues at stake. This progression from situations with relatively complete to relatively incomplete information also demonstrates what gaps in knowledge may exist and the consequences of those gaps. Of course, part of the value of going through an ecosystem services evaluation is to identify the gaps in existing information to show what types of research are needed.

Chapter 5 includes an extensive discussion of various implications and lessons learned from the case studies that are reviewed. These examples show that the ability to generate useful information about the value of ecosystem services varies widely across cases and circumstances. For some policy questions, enough is known about ecosystem service valuation to help in decision-making. As other examples make clear, knowledge and information may not yet be sufficient to estimate the value of ecosystem services with enough precision to answer policy-relevant questions. In general, the inability to generate relatively precise and reliable estimates of ecosystem values may arise from any combination of the following three reasons: (1)

insufficient ecological knowledge or information to estimate the quantity of ecosystem services produced or to estimate how ecosystem service production would change under alternative scenarios, (2) an inability of existing economic methods to generate precise estimates of value for the provision of various levels of ecosystem services, and (3) a lack of integration of ecological and economic analysis.

Studies that focus on valuing a single ecosystem service show promise of delivering results that can inform important policy decisions. In no instance, however, should the value of a single ecosystem service be confused with the value of the entire ecosystem. Unless it is clearly understood that valuing a single ecosystem service represents only a partial valuation of the natural processes in an ecosystem, such single service valuation exercises may provide a false signal of total value. Even when the goal of a valuation exercise is focused on a single ecosystem service, a workable understanding of the functioning of large parts or possibly the entire ecosystem may be required. Although the valuation of multiple ecosystem services is more difficult than the valuation of a single service, interconnections among services may make it necessary to expand the scope of the analysis. As noted previously, ecosystem processes are often spatially linked, especially in aquatic ecosystems. Full accounting of the consequences of actions on the value of ecosystem services requires understanding these spatial links and undertaking integrated studies at suitably large spatial scales to fully cover important effects. In generating estimates of the value of ecosystem services across larger spatial scales, extrapolation may be unavoidable, but it should be applied with careful scrutiny. Lastly, the value of ecosystem services depends upon underlying conditions. Ecosystem valuation studies should clearly present assumptions about underlying ecosystem and market conditions and how estimates of value could change with changes in these underlying conditions.

Building on the implications and lessons learned and on these preceding conclusions, the committee provides the following recommendations (Chapter 5):

- There is no perfect answer to questions about the proper scale and scope of analysis in ecosystem services valuation. One way to accomplish the integration of ecology and economics to value ecosystem services is to design the study to answer a particular policy question. The policy question then serves as the unifying frame that directs both ecological and economic analysis.
- Estimates of ecosystem value need to be placed in context. Assumptions about conditions in ecosystems outside the target ecosystem and assumptions about human behavior and institutions should be clearly specified.
- Concerted efforts should be made to overcome existing institutional barriers that prevent ready and effective collaboration among ecologists and economists regarding the valuation of ecosystem services. Furthermore, existing and future interdisciplinary programs aimed at integrated environmental analysis should be encouraged and supported.

## JUDGMENT, UNCERTAINTY, AND VALUATION

The valuation of aquatic and related terrestrial ecosystem services inevitably involves investigator judgments and some amount of uncertainty. Although unavoidable, uncertainty and the need to exercise professional judgment are not debilitating to ecosystem valuation. However, when such judgments are made it is important to explain why they are needed and to indicate the alternative ways in which judgment could have been exercised. It is also important that the sources of uncertainty be acknowledged, minimized, and accounted for in ways that ensure that a study's results and related decisions regarding ecosystem valuation are not systematically biased and do not convey a false sense of precision.

There are several cases in which investigators must use professional judgment in ecosystem valuation regarding how to frame a valuation study, how to address the methodological judgments that must be made during the study, and how to use peer review to identify and evaluate these judgments. Of these, perhaps the most important choice in any ecosystem valuation study is the selection of the question to be asked and addressed (i.e., "framing" the study). The case studies discussed in Chapter 6 illustrate the fact that the policy context unavoidably affects the framing of an ecosystem valuation study and therefore the type and level of analysis needed to answer it. Framing also affects the way in which people respond to any given issue. Analysts need to be aware of this and sensitive to the different ways of presenting data and issues, and should make a serious attempt to address all perspectives in their presentations because failure to do so could undermine the legitimacy of an ecosystem valuation study.

In most ecosystem valuation studies, an analyst will be called on to make various methodological judgments about how the study should be designed and conducted. Typically, these judgments will address issues such as whether, and at what rate, future benefits and costs should be discounted; whether to value goods and services by what people are willing to pay or what they would be willing to accept if these goods and services were reduced or lost; and how to account for and present distributional issues arising from possible policy measures. In many cases, different choices regarding some of these issues will make a substantial difference in the final valuation. The unavoidable need to make professional judgments in ecosystem valuation through choices of framing and methods suggests that there is a strong case for peer review to provide input on these issues before study design is complete and relatively unchangeable.

There are several major sources of uncertainty in the valuation of aquatic ecosystem services and several options for policymakers and analysts to respond. Model uncertainty arises for the obvious reason that in many cases the relationships between certain key variables are not known with certainty (i.e., the "true model" will not be known). Parameter uncertainty is one level below model uncertainty in the logical hierarchy of uncertainty in the valuation of ecosystem services. The almost inevitable uncertainty facing analysts involved in ecosystem valuation can be more or less severe depending on the availability of good probabilistic information or lack thereof (i.e., the amount of ambiguity). A favorable case would be one in which although there is uncertainty about some key magnitudes of various parameters, the analyst nevertheless has good probabilistic information. An alternative and common scenario in ecosystem valuation is one in which there is really no good probabilistic information about the likely magnitude of some variables, and what is available is based only on expert judgment. However, just as there are different types of uncertainty in ecosystem valuation, there are also different ways and decision criteria that an analyst can use to allow for uncertainty in the support of environmental decision-making; these are reviewed in Chapters 2 and 6. One of these is the use of Monte Carlo simulations as a method of estimating the range of possible outcomes and the parameters of its probability distribution. The outcome of an environmental policy choice under uncertainty is necessarily unpredictable, and risk aversion is a measure of what a person is willing to pay to avoid an uncertain outcome. In a heterogeneous population, the analyst will have to make an assumption about the level of risk aversion that is appropriate for the group as a whole.

Although considerable uncertainty exists about the value of ecosystem services, there is often the possibility of reducing this uncertainty over time through passive and/or active learning. Regardless of its source, the possibility of reducing uncertainty in the future through learning can affect current decisions, particularly when the impacts of those decisions are (effectively) irreversible (e.g., the construction or removal of a dam). With learning, there is an "option value" that needs to be incorporated into the analysis as part of the expected net benefits that reflects the value of the additional flexibility. This flexibility allows future decisions to respond to new information as it becomes available. It follows that in a cost-benefit analysis, measurement of the benefits of ecosystem protection through ecosystem valuation should consider the possibility of learning (i.e., should incorporate the option value). At present, only a limited amount of empirical work has been done on estimating the magnitude of option value. A natural extension of the observation that better decisions can be made if one waits for additional information is through the use of adaptive management. Adaptive management is a relatively new but increasingly used paradigm for confronting the inevitable uncertainty arising among management policy alternatives for large complex ecosystems or ecosystems in which functional relationships are poorly known. It provides a mechanism for learning systematically about the links between human societies and ecosystems, although it is not a tool for ecosystem valuation or a method of valuation per se.

Based on these conclusions, the committee makes the following recommendations regarding judgment and uncertainty in ecosystem valuation activities and methods and approaches to effectively and proactively respond to them (Chapter 6):

- Analysts must be aware of the importance of framing in designing and conducting ecosystem valuation studies so that the study is tailored to address the major questions at issue. Analysts should also be sensitive to the different ways of presenting study data, issues, and results and make a concerted attempt to address all relevant perspectives in their presentations.
- The decision to use WTP or WTA as a measure of the value of an ecosystem good or service is a choice about how an issue is framed. If the good or service being valued is unique and not easily substitutable with other goods or services, then these two measures are likely to result in very different valuation estimates. In such cases, the committee cannot reasonably recommend that the analyst report both sets of estimates in a form of sensitivity analysis because this may effectively double the work. Rather, the analyst should document carefully the ultimate choice made and clearly state that the answer would probably have been higher or lower had the alternative measure been selected and used.
- Because even small differences in a discount rate for a long-term environmental restoration project can result in order-of-magnitude differences in the present value of net benefits, in such cases the analyst should present figures on the sensitivity of the results to alternative choices for discount rates.
- Ecosystem valuation studies should undergo external review by peers and stakeholders early in their development when there remains a legitimate opportunity for revision of the study's key judgments.
- Analysts should establish a range for the major sources of uncertainty in an ecosystem valuation study whenever possible.

- Analysts will often have to make an assumption about the level of risk aversion that is appropriate for use in an ecosystem valuation study. In such cases, the best solution is to state clearly that the assumption about risk aversion will affect the outcome and to conduct sensitivity analyses to indicate how this assumption impacts the outcome of the study.
- There is a need for further research about the relative importance of and estimating the magnitude of option values in ecosystem valuation.
- Under conditions of uncertainty, irreversibility, and learning, there should be a clear preference for environmental policy measures that are flexible and minimize the commitment of fixed capital or that can be implemented on a small scale on a pilot or trial basis.

## ECOSYSTEM VALUATION: SYNTHESIS AND FUTURE DIRECTIONS

The final chapter of this report seeks to synthesize the current knowledge regarding ecosystem valuation in a way that will be useful to resource managers and policymakers as they incorporate the value of ecosystem services into their decisions. A synthesis of the report's general premises and major conclusions regarding ecosystem valuation suggests that a number of issues or factors enter into the appropriate design of a study of the value of aquatic ecosystem services. The context of the study and the way in which the resulting values will be used play a key role in determining the type of value estimate that is needed. In addition, the type of information that is required to answer the valuation question and the amount of information that is available about key economic and ecological relationships are important considerations. This strongly suggests that the valuation exercise will be very context specific and that a single, "one-size-fits-all" or "cookbook" approach cannot be used. Instead, the resource manager or decision maker who is conducting a study or evaluating the results of a valuation study must assess how well the study is designed in the context of the specific problem it seeks to address. In this regard, Chapter 7 provides a checklist to aid in this assessment that identifies questions that should be openly discussed and satisfactorily resolved in the course of the valuation exercise.

Finally, Chapter 7 identifies what the committee feels are the most pressing recommendations for improving the estimation of ecosystem values and their use in decisions regarding ecosystem protection, preservation, or restoration. These overarching recommendations are based on, and in some cases build on, the more specific recommendations presented at the ends of the previous chapters; they include (1) overarching recommendations for conducting ecosystem valuation and (2) overarching research needs, which imply recommendations regarding future research funding.

# 1
# Introduction

The biota and physical structures of ecosystems provide a wide variety of marketable goods—fish and lumber being two familiar examples. Moreover, society is increasingly recognizing the myriad life support functions, the observable manifestations of ecosystem processes that ecosystems provide and without which human civilizations could not thrive (Daily, 1997; Naeem et al., 1999). These include water purification, recharging of groundwater, nutrient recycling, decomposition of wastes, regulation of climate, and maintenance of biodiversity. Derived from the physical, biological, and chemical processes at work in natural ecosystems, these functions are seldom experienced directly by users of the resource. Rather, it is the services provided by the ecosystems—services that create value for human users, such as flood risk reduction and water supply—together with the ecosystem goods, that are the subject of this report.

Despite the importance of ecosystem functions and services, they are often overlooked or taken for granted and their value implicitly set at zero in decisions concerning conservation or restoration (Bingham et al., 1995; Heal, 2000; Postel and Carpenter, 1997). Choices between the conservation and restoration of ecosystems and the continuation and expansion of human activities have to be made however in the recognition of conflicts between the expansion of certain human activities and the continued provision of valued ecosystem goods and services. In making these choices, the economic values of ecosystem goods and services should be assessed and compared with the economic values of activities that may compromise them. Although factors other than economic values may ultimately enter into the choices, these values are important inputs to the environmental policy decision-making process.

Aquatic ecosystems include freshwater, marine, and estuarine surface waterbodies. These incorporate lakes, rivers, streams, coastal waters, estuaries, and wetlands, together with their associated flora and fauna. Each of these entities is connected to a greater ecological and hydrological landscape that includes adjacent riparian areas, upland terrestrial ecosystems, and underlying groundwater aquifers. As discussed in detail in Chapter 3, the term "aquatic ecosystems" used in this report includes related terrestrial ecosystems and underlying aquifers.

Historically, the United States had an abundance of aquatic ecosystems. However many of these systems have been lost altogether, or the species of plants and animals they support have been diminished in kind and number. For example, between the time of European settlement and about 1950, it is estimated that more than half of the nation's wetlands were converted for agricultural or other land uses(Heinz Center, 2002; NRC, 2001). An additional 10 percent of the wetlands remaining in 1950 have since been converted to another use (see also Table 1-1). In addition, less than 2 percent of the nation's 3.1 million miles of rivers and stream remain free flowing for longer than 125 miles and include more than 75,000 dams larger than 6 feet and 2.5 million smaller dams (TNC, 1998). Within the United States, more than 60 percent of freshwater mussels and crayfish are considered rare or imperiled and 35 percent or more of fish and aquatic amphibian species are at some risk of extinction (Abell et al., 2000). Thus, the number and amount of intact functional aquatic ecosystems have been substantially reduced in recent

TABLE 1-1 Recent Wetland Losses in the United States

| Period | Losses Due to Agriculture | Losses Due to Non-Agriculture[a] | Total Acreage Lost[b] (Annual Average Loss) |
|---|---|---|---|
| Mid-1970s to mid-1980s (10 years) | 137,540 acres per year (54% of loss) | 117,230 acres per year (46% of loss) | 2,547,700 acres (254,770 acres per year) |
| 1986 to 1997 (11 years) | 15,222 acres per year (26% of loss) | 43,324 acres per year (76% of loss) | 644,000 acres (58,545 acres per year) |

SOURCE: Adapted from Dahl (2000); Dahl and Johnson (1991); NRC (2001).
[a] Non-agricultural losses include those from silviculture, urban, and rural development uses.
[b] Total acreage lost was determined by multiplying the annual acreage loss by the total number of years in that time period.

decades. This relative scarceness has called increasing attention to the need to better understand the functionality and value of the remaining ecosystems to society.

Despite the large losses and changes in these systems, aquatic ecosystems remain broadly and heterogeneously distributed across the nation. At a glance, there are almost 4 million miles of rivers and streams, 59,000 miles of ocean shoreline waters, and 5,500 miles of Great Lakes shoreline in the United States (EPA, 2002). There are 87,000 square miles of estuaries, while lakes, reservoirs, and ponds account for more than 40 million acres. As of 1997, the lower 48 states contained about 165,000 square miles (105.5 million acres) of wetlands of all types—an area about the size of California (Dahl, 2000). Figure 1-1 shows major rivers and streams. Figure 1-2 shows major aquifers in the United States classified by major features that affect the occurrence and availability of groundwater. A variety of federal programs report on the extent, status, and related trends of aquatic ecosystems located throughout the United States. Although it is beyond the scope of this report to review systematically or even summarize all such programs, a few of the largest and most important programs are described briefly in Chapter 3.

As noted above, aquatic ecosystems collectively perform numerous interrelated functions and provide a wide range of services. In addition, many aquatic ecosystems support the economic livelihood of local communities through commercial fishing and by serving the recreational sector. To illustrate the importance of these activities, recreational fishing alone generated an estimated $116 billion in total economic output the United States in 2001 (American Sportsfishing Association, 2002). The continuance or growth of these types of economic activities is directly related to the extent and health of these natural ecosystems. However, human activities and rapid population growth (often preferentially in or near aquatic ecosystems), along with historical and ongoing industrial, commercial, and residential development, have led to increased pollution, adverse modification, and destruction of remaining (especially pristine) aquatic ecosystems (Baron et al., 2003; Carpenter, et al., 1998; Howarth et al., 2000; NRC, 1992). At the same time, increased human demand for water has reduced the amount available to support these ecosystems (Heinz Center, 2002; Jackson et al., 2001).

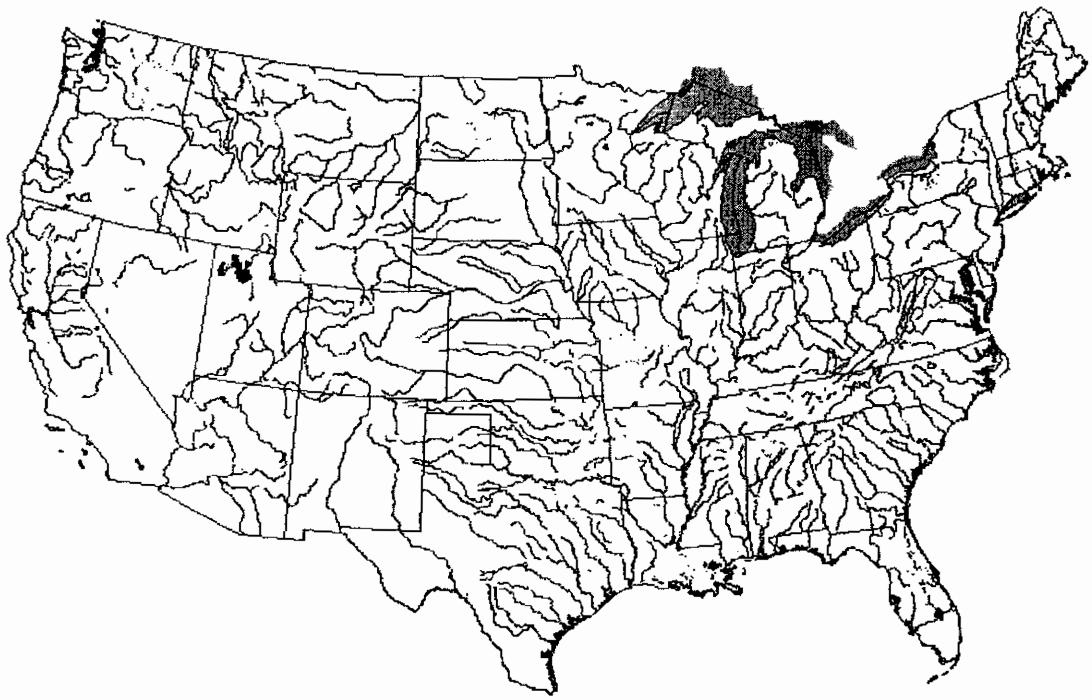

FIGURE 1-1 Major rivers and streams of the co terminous United States. SOURCE: Generated from the National Atlas of the United States (available on-line at *http://www.nationalatlas.gov*).

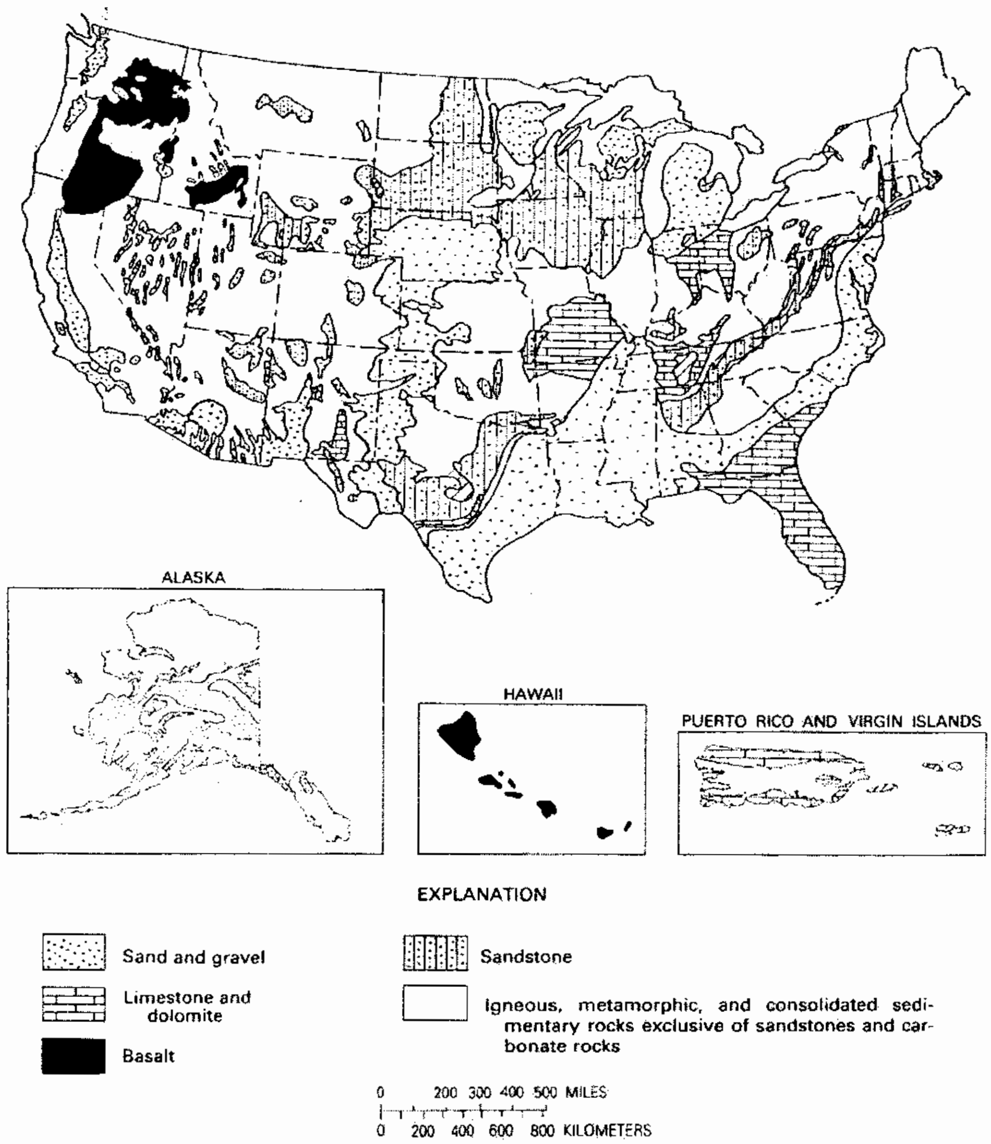


FIGURE 1-2 Groundwater regions in the United States. Note: Shading refers to principal types of water-bearing rocks. SOURCE: Heath (1984).

In the case of commercial and recreational fishing, pollution of aquatic ecosystems has adversely affected annual fish catch. For example, coastal areas and estuaries provide important nurseries for many species of commercially valuable fish and shellfish and have been adversely affected by nutrient pollution and habitat loss (Beck et al., 2001, 2003). Moreover, increasing demand for the services of aquatic ecosystems has resulted in a huge increase in the raising of fish (aquaculture) worldwide, which itself is having substantive effects on natural aquatic ecosystems (Naylor, 2001). This has occurred despite an increase in federal, state, and local regulations intended to restore and protect these natural resources. In this regard, many of the regulatory efforts to control pollution stem from the Clean Water Act (CWA),[1] which originally focused on controlling point source pollution and limiting the destruction of wetlands.

Initially, certain large point sources of pollution were exempted from this federal act, such as concentrated or confined animal feeding operations (CAFOs), which have been responsible for pollution of a number of important aquatic ecosystems. However, CAFOs have recently been required to meet tighter discharge standards (EPA, 2003a) under the CWA. At present, nonpoint source (NPS) pollution is widely considered the leading remaining cause of water quality problems throughout much of the United States. The sources of NPS pollution to aquatic ecosystems are varied and range from runoff of fertilizers and pesticides applied to farm fields to atmospheric deposition of rainfall polluted from automobile emissions (Carpenter et al., 1998; Howarth et al., 2002).

This chapter serves as an introduction to the extent and importance of aquatic and related terrestrial ecosystems throughout the United States. It provides a statement of the problem of attempting to assess and value the services of aquatic and related ecosystems, summarizes the origin and scope of the study, and describes the perspective of the committee and this report. Chapter 2 provides an overview of the different sources and meanings of "value" in the policy process with a focus on economic valuation and the role it can play in improving environmental decision-making. Chapter 3 reviews some existing definitions of aquatic and related terrestrial ecosystems; describes their associated structures and functions; and introduces their translation to ecosystem goods and services. Chapter 4 provides a review of key existing methods of nonmarket valuation for aquatic ecosystems and issues related to their development and successful application. Chapter 5 focuses on translating ecosystem functions into services using an extensive series of case studies that compare and contrast such efforts in order to develop "lessons learned" that can be applied in future ecosystem valuation activities. Chapter 6 assesses judgment and uncertainty associated with ecosystem valuation and suggests how analysts and decision-makers can and should respond. Lastly, Chapter 7 synthesizes the current knowledge regarding ecosystem valuation and builds on the preceding chapters in order to provide guidelines for policymakers and planners concerned with the management, protection, and restoration of aquatic ecosystems. It also identifies what the committee feels are overarching recommendations for improving the valuation of ecosystem services and related research needs.

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[1] Growing public awareness of and concern for controlling water pollution nationwide led to enactment of the Federal Water Pollution Control Act (FWPCA; enacted in 1948) Amendments of 1972. The Clean Water Act, as it became known, arose from 1977 amendments to the FWPCA and is a comprehensive statute intended to restore and maintain the chemical, physical, and biological integrity of the nation's waters. To accomplish this national objective, the CWA seeks to attain a level of water quality that "provides for the protection and propagation of fish, shellfish, and wildlife, and provides for recreation in and on the water." Primary authority for implementation and enforcement of the CWA—which has been amended almost yearly since its inception—rests with the U.S. Environmental Protection Agency.

## STATEMENT OF THE PROBLEM

Some believe that environmental amenities and services lie outside the scope of economic analyses, arguing that the need to protect environmental assets is self-evident and not properly the subject of economic analyses (see Chapter 2 for further discussion). However, wherever there is scarcity and the need to choose between alternatives, the question of relative values is unavoidable. It may be costly to protect, conserve, and restore aquatic ecosystems, and the costs are borne by giving up benefits in other parts of the economy, now or in the future. When ecosystem protection projects and policies are proposed, it is appropriate to ask whether they achieve the stated goals in a cost-effective and efficient manner, whether the costs are commensurate with the benefits received, what society's costs are if protection is not provided, and whether costs and benefits are properly allocated across the present population and across generations.

Economic valuation requires that ecosystems be described in terms of the goods and services they provide to humans or other beneficiaries. Goods and services, in turn, must be quantified and measured on a common (though not necessarily monetary) scale if improvements to one ecosystem are to be compared to improvements to another. Although the issues that this raises apply to all types of ecosystems, the use of such information has started to come into particularly sharp focus for aquatic ecosystems and especially for wetlands (NRC, 2001).

Studying ecosystem services presents several challenges that are discussed throughout this report. The most fundamental challenge lies in providing an explicit description of the links between the structure and function of natural systems and the benefits (i.e., goods and services) derived by humanity. This problem is complicated by the fact that humans are an integral part of the system; by incomplete knowledge of how ecosystems function; and by the fact that ecosystem services tend to be specific to locations and situations, thus making it difficult to develop generic principles or identify generic characteristics.

The challenges to both ecologists and economists implicit in valuing ecosystem services are summarized in Figure 1-3. Human actions affect the structure, functions, and goods and

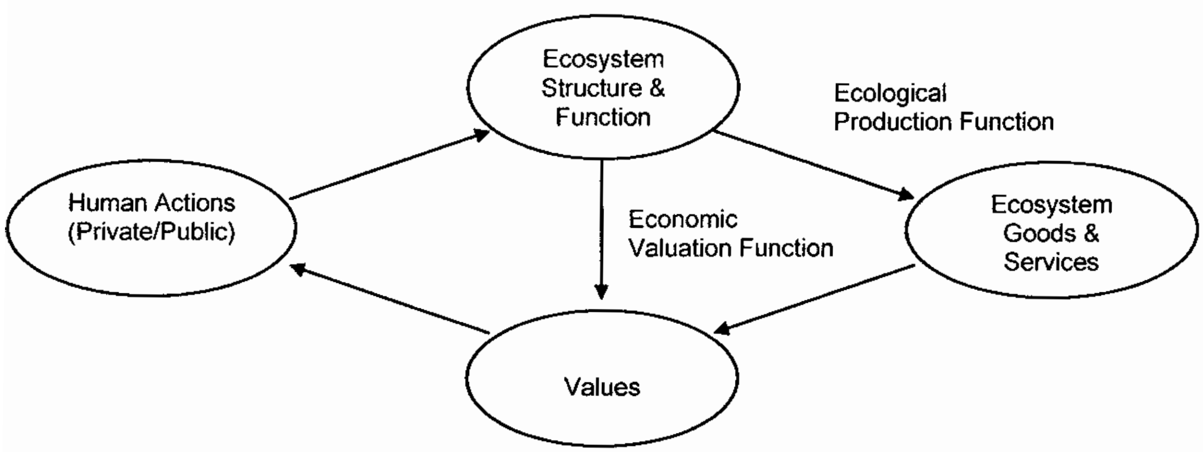


FIGURE 1-3 Components of ecosystem valuation: ecosystem structure and function, goods and services, human actions (policies), and values (see Figure 7-1 for an expanded version of this figure).

services of ecosystems. Ecosystem conditions are also affected by various biophysical parameters (not shown in figure). The translation from ecosystem structure and functions to ecosystem goods and services is given by an ecological production function, and the translation from ecosystem goods and services to value is given by an economic valuation function. There may be occasions in which the structure of the ecosystem is valued directly by humans, without the intermediation of functions, goods, or services. For example, people may value the existence redwood forests in their own right rather than because of any functions, goods, or services that they might provide; a possibility indicated in Figure 1-3 by the direct connection from ecosystem structure to values (also given by an economic valuation function). Estimating the value of ecosystem services requires uncovering both the ecological production function and the economic valuation function. As Chapters 3, 4, and 5 illustrate, uncovering each of these functions is difficult. Furthermore, because aquatic ecosystems are complex, the production of goods and services can be complicated and indirect; this in turn makes the translation from ecosystem structure and function to ecosystem goods and services difficult. The lack of markets and market prices and of other direct behavioral links to underlying values makes the translation from quantities of goods and services to value difficult as well.

Although valuing ecosystem services does not require knowledge of the function that maps human actions into ecosystem conditions, evaluating whether certain actions are in society's best interest does require this knowledge. For example, knowing whether to allow housing development in a watershed or timber harvesting in a forest patch requires predictions of how these actions will perturb ecosystems. This perturbation will change the production and value of ecosystem goods and services, and can then be compared to the direct economic value generated by the action (e.g., housing values, value of timber harvest) to see whether or not the action generates positive net benefits.

Where an ecosystem's goods and services can be identified and measured, it will often be possible to assign values to them by employing existing economic valuation methods. Chapter 4 provides a summary of key existing nonmarket valuation methods for (primarily aquatic) ecosystem services. Some ecosystem goods and services cannot be valued because they are not quantifiable or because available methods are not appropriate or reliable. In other cases, the cost of valuing a particular service may rule out the use of a formal method. Available economic valuation methods are complex and demanding. The results of applying these methods may be subject to judgment and uncertainty and must be interpreted with caution. Still, the general sense of a very large literature on the development and application of various methods is that they are relatively well evolved and capable of providing useful information in support of improved ecosystem valuation. There is little to be gained from a comprehensive National Academies review of these valuation methods. Indeed, the literature contains numerous authoritative reviews and critiques, and some federal agencies have published their own assessments and guidelines, which are cited and discussed briefly in Chapter 4. Thus, an important question for this committee was not how to use any particular valuation method, but how to address ecosystem services for which no existing valuation method has been identified, and how to integrate economic and ecological analysis to obtain economic values of ecosystem conservation. Similarly, while not repeating existing reviews or assessments of valuation methods, this report addresses the decision-making consequences of judgment and uncertainty, including the implications for the selection of methods in specific applications.

Probably the greatest challenge for successful valuation of ecosystem services is to integrate studies of the ecological production function with studies of the economic valuation function. After all, an understanding of the goods and services provided by a particular ecological resource, the interactions among them, and their sustainable levels can come only from ecological research and models. To integrate economic and ecological studies, the definitions of ecosystem goods and services must match across studies. In other words, the quantities of goods and services must be defined in a similar manner for both ecological studies and economic valuation studies. Failure to do so means that the results of ecological studies cannot be carried over into economic valuation studies. Attempts to value ecosystem services without this key link will either fail to have ecological underpinnings or fail to be relevant as valuation studies.

Although there has been great progress in ecology in improving our understanding of aquatic ecosystem structure and function and in economics in developing and applying nonmarket valuation techniques, there remains a gap between the two. There are few examples of studies that have successfully translated knowledge about ecosystems into a form where economic valuation can be applied in a meaningful way. Several factors contribute to this continued lack of integration. First, some ecologists and economists hold vastly different views on the current "state of the world" and the direction in which it is headed. More recently, however, there has been mutual recognition among at least some ecologists and economists that addressing issues such as conserving ecosystems and biodiversity requires the input of both disciplines to be successful. A second reason for the lack of integration is that ecology and economics are separate disciplines, one in natural science and the other in social science. The traditional academic organization and the reward structure for scientists often make collaboration across disciplinary boundaries difficult even when the desire to do so exists (e.g., Bingham et al., 1995). Third, the ecosystem services paradigm is relatively new, as are attempts to value ecosystem services. Building the necessary working relationships and integrating methods across disciplines will take time.

Integrated studies of the value of ecosystem goods and services are now emerging. Chapter 5 reviews several such studies, beginning with situations in which the focus is on valuing a single ecosystem service, progressing to attempts to value multiple ecosystem services, and ending by reviewing analyses that attempt to encompass all services produced by an ecosystem. In some cases, it may be possible to generate relatively precise estimates of value; in other cases, all that may be possible is a rough categorization ("a lot" versus "a little"). Whether this is sufficient information depends on the circumstances. In some instances, a rough estimate may be sufficient to decide that one option is preferable to another, whereas tougher decisions will require more refined information. This progression from situations with good to poor information also demonstrates what types of information will often be lacking and the consequences of those gaps. Indeed, part of the value of going through an ecosystem services evaluation is to point out the gaps in existing information and show what research is needed to fill these gaps.

It is clear that more categories of human endeavor will in the future be evaluated to some extent in terms of environmental effects and impacts on quality of life. The emerging desire to measure the environmental costs of human activities, or to assess the benefits of environmental protection and restoration, has challenged the state of the art in environmental evaluation in both the ecological and the social sciences. From an ecological perspective, the challenge is to interpret basic research on ecosystem functions so that service-level information can be communicated to economists. For economics and related social sciences, the challenge is to identify the values of both tangible and intangible goods and services associated with ecosystems and to address the problem of decision-making in the presence of partial valuation. The combined challenge is to develop and apply methods to assess the values of human-induced changes in aquatic and related terrestrial ecosystem functions and services.

## STUDY ORIGIN AND SCOPE

This study was conceived in 1997 at a strategic planning session of Water Science and Technology Board (WSTB) of the National Research Council (NRC). Initially, the NRC organized and hosted a planning workshop to assess the feasibility of and need for an NRC study of the functions and associated economic values of aquatic ecosystems. Fourteen key experts involved or interested in the management, protection, and restoration of aquatic ecosystems—including representatives of the study sponsors, the U.S. Army Corps of Engineers (USACE), U.S. Environmental Protection Agency (EPA), and U.S. Department of Agriculture (USDA)—participated in the workshop that was held early in November 1999 in Washington, D.C. All participants agreed that an NRC study of valuation methods used to assess aquatic ecosystem services, rather than functions, was feasible and timely and would make a significant contribution toward advancing the understanding and appropriate use of economic valuation methods in environmental decision-making. However, it is important to note that the NRC has released several reports in the last decade that are somewhat related to this study. These are listed and briefly summarized in ascending chronological order in Appendix A. Furthermore, there has been a general increase in interest in the area of economic valuation of ecosystem services and its role in environmental policy and decision-making since the committee was formed in early 2002 (discussed below). For example, the EPA's Science Advisory Board (SAB) recently established a panel to review EPA's draft Environmental Economics Research Strategy (EPA, 2003b).[2]

The WSTB developed a full study proposal and while several minor changes were made to the proposal in response to the sponsoring (and nonsponsoring) agencies, one significant change was made. As a compromise to the USACE's desire to expand the scope of the study to include all ecosystems, it was decided and subsequently agreed by the NRC and all study sponsors to expand the study proposal to include "related terrestrial ecosystems." The original basis for this change in language and study focus was the key 1983 water resources planning report *Economic and Environmental Principles and Guidelines for Water and Related Land Resources Implementation Studies* (WRC, 1983). The implications of linking "related terrestrial ecosystems" to aquatic ecosystems are discussed more fully in Chapter 3.

The committee's statement of task (see Box ES-1) was to evaluate methods for assessing the economic value of the goods and services provided by aquatic and related terrestrial ecosystems. More specifically, it asks "What lessons can be learned from a comparative review of past attempts to value ecosystem services—particularly, are there significant differences between eastern and western U.S. perspectives on these issues?" As is evident throughout this report, the committee made extensive use of case studies in ecosystem services valuation (especially in Chapter 5) to help develop many of its conclusions and recommendations and

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[2] The panel consists of members of the existing SAB Environmental Economics Advisory Committee to which several experts were added (including several members of this NRC committee) to form the Advisory Panel on the Environmental Economics Research Strategy (see *http://www.epa.gov/sab/pdf/apeers_bios_for-web.pdf* and *http://es.epa.gov/ncer/events/news/2003/06_23_03a.html* for further information).

respond to this and other elements of the statement of task. Although the case studies are drawn primarily from throughout the United States, including eastern and western areas, the committee decided early in its deliberations that it would not make geographic distinctions in developing implications and lessons learned from the case studies.

This report is about placing values on the goods and services that ecosystems provide to human societies, with its principal focus on the goods and services provided by aquatic and related terrestrial ecosystems. Furthermore, the report focuses on freshwater and estuarine systems, eschewing extensive consideration of marine and groundwater systems. This reflects an intentional effort to focus on management and valuation issues confronting state and federal agencies for these ecosystems. However, because the principles and practices of valuing ecosystem goods and services are rarely sensitive to whether the underlying ecosystem is aquatic or terrestrial, the report's various conclusions and recommendations are likely to be directly or at least indirectly applicable to the valuation of the goods and services provided by any ecosystem.

## PERSPECTIVE OF THIS REPORT

Several elements are fundamental to the perspective taken by the committee as it developed this report. The first is that ecosystems provide goods and services, sometimes very important ones, to society (see for example, Daily, 1997; de Groot et al., 2002; Ewel, 2002; Peterson and Lubchenco, 2002; Postel and Carpenter, 1997). The second element is that in many cases these goods and services can be quantified and an economic value can be placed on them. In large part, the remaining chapters discuss how to do this. A third element is that economic valuation can often be useful in support of environmental policy decision-making. Although the economic value of an ecosystem may not capture all of the reasons it is valued and conserved, economic valuation captures some of these reasons—perhaps most of them under certain circumstances. This valuation, in turn, becomes a necessary input to decisions about environmental conservation, particularly in situations where there is an apparent conflict between conservation or restoration and a conventional idea of economic progress, as indicated by gross national or state product measured at market prices.

In many cases, some reviewed in the following chapters, careful valuation shows that conservation is economically beneficial, whereas the destruction or modification of natural systems is economically harmful. Finally, the concept of economic value is very inclusive, much more so than is recognized and appreciated outside the economics profession. Consequently, many of what noneconomists typically consider to be noneconomic values are in fact captured (at least to some extent) by economists' estimates of value—especially by what is called "existence value."

The reason economic valuation is more comprehensive than generally recognized is that economists recognize two basic types of value, use and nonuse values (see Chapters 2 and 4 for a more complete discussion). In brief, use values are those that derive from using a good or service provided by an ecosystem, such as using a lake for fishing or swimming, lake water for drinking or irrigation, or an estuary for boating. On the other hand, an example of a type of nonuse value is an existence value; a person may value the existence of a species even though he or she will never make any *use* of this species or of any of its members. Existence values, although often difficult and controversial to measure, are legitimate and indeed important economic values since people are willing to pay (see more below) for the continued existence of species or landscapes. Existence values also affect the way people behave, and anything that changes behavior has economic consequences. For example, even if people are not able to pay directly for the preservation of a species, the value they place on it might affect other aspects of their behavior, such as how they vote or their choice of products in the market. Values that lead to behavior changes are therefore economic values, even though their origins may lie in ethical, aesthetic, or religious beliefs (see Chapter 2 for further information). However, there could be occasions on which people value ecosystems, but this value is not reflected in any change in their behavior and is never revealed. For example, they might for some reason wish to keep their valuation secret. In such a case, economic methods of measuring values would fail to reflect a person's valuation.

Valuation studies may be conducted in many different contexts, and the context can affect some aspects of the study. A study may be conducted as part of a policy analysis, as in the case of the restoration of the New York Catskills watershed, or in the context of environmental litigation related to the *Exxon Valdez* oil spill (see Chapter 5). Alternatively, a valuation study may be conducted in the context of a NRDA (natural resource damage assessment) required by the federal Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).[3] As can be seen in the case studies developed in later chapters, the context can have an impact on the way a valuation study is framed (see Chapters 2 and 6) and on the way it is developed.

## SUMMARY AND CONCLUSIONS

Aquatic and related terrestrial ecosystems are broadly distributed across the nation, perform numerous interrelated functions, and provide a wide range of important goods and services. In addition, many aquatic ecosystems enhance the economic livelihood of local communities by supporting commercial fishing, supporting agriculture, and serving the recreational sector. The continuance or growth of these types of economic activities is directly related to the extent and health of these natural ecosystems. However, human activities, rapid population growth, and industrial, commercial, and residential development have all led to increased pollution, adverse modification, and destruction of remaining aquatic ecosystems—despite an increase in federal, state, and local regulations intended to protect, conserve, and restore these natural resources. Increased human demand for water has simultaneously reduced the amount available to support these ecosystems.

Despite growing recognition of the importance of ecosystem functions and services, they are often taken for granted and overlooked in environmental decision-making. Thus, choices between the conservation and restoration of some ecosystems and the continuation and expansion of human activities in others have to be made with an enhanced recognition of this potential for conflict. In making these choices, the economic values of these ecosystem goods and services to society have to be known, so that they can be compared with the economic values of activities that may compromise them and improvements to one ecosystem can be compared to

[3] In response to growing public concern over health and environmental risks posed by hazardous waste sites, Congress enacted CERCLA, commonly known as the Superfund program, in 1980 to identify and clean up such sites. Superfund is administered by EPA in cooperation with individual sites throughout the United States; and further information can be found at *http://www.epa.gov/superfund/action/law/cercla.htm.*

those in another.

The fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of links between the structures and functions of natural systems and the benefits (i.e., goods and services) derived by humanity and is summarized in Figure 1-3. Ecosystems are complex however, making the translation from ecosystem function to ecosystem goods and services (i.e., the ecological production function) difficult. Similarly, the lack of markets and market prices and of other direct behavioral links to underlying values makes the translation from quantities of goods and services to value (i.e., the economic valuation function) quite difficult.

Probably the greatest challenge for successful valuation of ecosystem services is to integrate studies of the ecological production function with studies of the economic valuation function. To do this, the definitions of ecosystem goods and services must match across studies. Failure to do this means that the results of ecological studies cannot be carried over into economic valuation studies. Attempts to value ecosystem services without this key link will either fail to have ecological underpinnings or fail to make be relevant as valuation studies.

Where an ecosystem's services and goods can be identified and measured, it will often be possible to assign values to them by employing existing economic (primarily nonmarket) valuation methods. Some ecosystem goods and services cannot be valued because they are not quantifiable or because available methods are not appropriate or reliable; in other cases, the cost of valuing a particular service may rule out the use of a formal method. Economic valuation methods are complex and demanding, and the results of applying these methods may be subject to judgment, uncertainty, and bias and must be interpreted with caution. However, based on an assessment of a very large literature on the development and application of various economic valuation methods, the committee concludes that they are relatively mature and capable of providing useful information in support of improved environmental decision-making.

Although there has been great progress in ecology in better understanding ecosystem structure and functions, and in economics in developing and applying nonmarket valuation techniques, there remains a gap between the two. The challenge from an ecological perspective is to interpret basic research on ecosystem functions so that service-level information can be communicated to economists. The challenge for economics and related social sciences is to identify the values of both tangible and intangible goods and services associated with ecosystems while addressing the problem of decision-making in the presence of partial valuation. The combined challenge is to develop and apply methods to assess the values of human-induced changes in aquatic and related terrestrial ecosystem functions and services.

Lastly, this report is primarily concerned with valuing the goods and services that aquatic and related terrestrial ecosystems provide to human societies. However, because the principles and practices of valuing ecosystem goods and services are rarely sensitive to whether the underlying ecosystem is strictly aquatic or terrestrial, many of its conclusions and recommendations are likely to be directly or at least indirectly applicable to the valuation of goods and services provided by any ecosystem.

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# 2
# The Meaning of Value and Use of Economic Valuation in the Environmental Policy Decision-Making Process

## INTRODUCTION

In developing a perspective and providing expert advice on valuing aquatic and related terrestrial ecosystems, it is necessary to begin with a clear discussion and statement of what it means to value something and of the role of "valuation" in environmental policy decision-making. Environmental issues and ecosystems have been at the core of many recent philosophical discussions regarding value (e.g., Goulder and Kennedy, 1997; Sagoff, 1997; Turner, 1999). Fundamentally, these debates about the value of ecosystems derive from two points of view. One view is that some values of ecosystems and their services are non-anthropocentric —that nonhuman species have moral interests or value in themselves. The other view, which includes the economic approach to valuation, is that all values are anthropocentric.

While acknowledging the potential validity of the first point of view, the committee was charged (see Chapter 1 and Box ES-1) specifically with assessing methods of valuing aquatic and related terrestrial ecosystems using economic methods, an approach that views values as inherently anthropocentric. For that reason, this report focuses on the sources of ecological value that can be captured through economic valuation.[1] However, the committee recognizes that all kinds of value may ultimately contribute to decisions regarding ecosystem use, preservation, or restoration. The committee's approach is consistent with the approach taken in the international Millennium Ecosystem Assessment,[2] which focuses on contributions of ecosystems to human well-being while at the same time recognizing that potential for non-anthropocentric sources of value.

Although this report focuses on the subset of values that can be captured through economic valuation, it is important to emphasize that this subset of values is quite broad; indeed, it is much broader than is often presumed. There are many misconceptions about the term "economic valuation." For example, many believe that the term refers simply to an assessment of the commercial value of something. In fact, the economic view of value actually includes many components that have no commercial or market basis (Freeman, 1993a; Krutilla, 1967), such as the value that individuals place on the beauty of a natural landscape or the existence of a species that has no commercial value. Thus, although economic valuation does not include all sources of value that have been identified or that are potentially important, it encompasses a very broad array of values. In addition, it provides a systematic way in which those values can be factored into environmental policy choices. This chapter provides an overview of economic

[1] Unless otherwise noted, use of the terms "value," "valuing," or "valuation" in this report refers to economic valuation; more specifically, the economic valuation of ecosystem goods and services.

[2] The Millennium Ecosystem Assessment was launched in June 2001 to help meet the needs of decision-makers and the public for scientific information concerning the consequences of ecosystem change for human well-being and options for responding to such changes (see Chapter 3 and *http://www.millenniumassessment.org/en/index.aspx* for further information).

valuation and the role it can play in improving environmental decision-making. The purpose is first to identify the values that are, and those that are not, captured by the economic approach to valuation and then to discuss how a quantification of the values can contribute to better environmental decision-making.

The chapter is divided into two main sections. The first discusses the role of economic valuation in the policy process and addresses the different meanings and sources of value in this context. The role and importance of quantifying values are discussed next, followed by a discussion of how information about values can be used in policy decisions. Finally, the importance of "framing" the valuation question appropriately is discussed, since the way in which a valuation exercise is defined can have a significant impact on the results that emerge from it.

Given this overview, the following section provides a more detailed examination of economic valuation. The section begins with a description of the "total economic value" framework, from which it is clear that economic valuation includes a wide array of values—many (in some cases most) of which are unrelated to any market or commercial value. This is followed by a discussion of quantifying value using a monetary metric. Two monetary metrics are described, willingness to pay (WTP) and willingness to accept (WTA), and the implications of using one versus the other are discussed. Finally, a discussion of discounting follows because many environmental policy impacts extend over long durations and it is important to incorporate the timing of these impacts into any valuation analysis. Discounting is the approach most commonly used in economic analysis to capture the timing of benefits and costs. The important distinction between discounting as a means of weighing the utility of future generations differently from that of present generations (utility discounting) and discounting as a means of weighing consumption (through benefits and costs) differently at different times (consumption discounting) is highlighted. This chapter closes with a summary of its conclusions and recommendations.

The broad overview of economic valuation provided in this chapter is followed in subsequent chapters by more detailed discussions of the types of ecosystem services that can be valued, the economic methods that can currently be used to quantify those values, and the role of professional judgment and uncertainty in ecosystem valuation.

## ROLE OF ECONOMIC VALUATION

### Different Sources and Meanings of Value

Given the crucial role that ecosystems and their services play in supporting human, animal, plant, and microbial populations, there is now widespread agreement that ecosystems are "valuable" and that decision-makers ranging from individuals to governments should consider the "value" of these ecosystems and the services they provide to society (Daily, 1997). However, there are different views on what this means and on the sources of that value. The literature on environmental philosophy and ethics distinguishes between (1) *instrumental* and *intrinsic* values, (2) *anthropocentric* and *biocentric* (or *ecocentric*) values, and (3) *utilitarian* and *deontological* values (Callicott, 2004). In order to place economic valuation in the context of these distinctions, each is discussed briefly below.

The instrumental value of an ecosystem service is a value derived from its role as a means toward an end other than itself. In other words, its value is derived from its usefulness in achieving a goal. In contrast, intrinsic value is the value that exists independently of any such contribution; it reflects the value of something for its own sake. For example, if a fish population provides a source of food for either humans or other species, it has instrumental value. This value stems from its contribution to the goal of sustaining the consuming population. If it continued to have value even if it were no longer "useful" to these populations (e.g., if an alternative, preferred food source were discovered), that remaining value would be its intrinsic value. For example, if the Grand Canyon and the Florida Everglades have intrinsic value, that component of value would be independent of whether humans directly or indirectly use them—either as sites for recreation, study or even contemplation. Intrinsic value can also stem from heritage or cultural sources, such as the value of culturally important burial grounds. Because intrinsic value is the value of something unrelated to its instrumental use of any kind, it is often termed "noninstrumental" value.

Anthropocentricism assumes that only human beings have intrinsic value and that the value of everything else is instrumental to human goals. To say that all values are anthropocentric, however, assumes that only humans assign value, and thus the value of other organisms stems from their usefulness to humans. Non-anthropocentric or biocentric values assume that certain things have value even if no human being thinks so. Thus, a biocentric approach assigns intrinsic value to all individual organisms, including but not limited to humans. Within this framework, intrinsic value or worth reflects more than humans caring about nonhumans and includes, in addition, the recognition that nonhumans have worth or value that is independent of any human caring or any satisfaction humans might receive from them. For example, a biocentric approach would assign a positive value to an obscure fish population (e.g., the snail darter; see more below) even if no human being feels that it is valuable and thus worth preserving. Clearly, both instrumental value and intrinsic value can be either anthropocentric or non-anthropocentric (see Callicott, 2004; Turner, 1999).

Intrinsic value is related but not identical to what economists call "existence value," which reflects the desire by some individuals to preserve and ensure the continued existence of certain species or environments. Existence value is an anthropocentric and utilitarian concept of value. Utilitarian values stem from the ability to provide "welfare," broadly defined to reflect the overall well-being of an individual or group of individuals. In this sense, utilitarian values are instrumental in that they are viewed as a means toward the end result of increased human welfare as defined by human preferences, without any judgment about whether those preferences are "good" or "bad." Existence values still stem from the fact that continued existence generates welfare for those individuals, rather than from the intrinsic value of nonhuman species. As such, there is the potential for substitution or replacement of this source of welfare with an alternative source (i.e., more of something else). In fact, implicit in the economic definition of existence values is the possibility of a welfare-neutral trade-off between continued existence of the species or environment and other things that also provide utility (see more detailed discussion below). Thus, the utilitarian approach implicitly assumes that existence value is an anthropocentric instrumental value that is potentially substitutable.[3]

In contrast, under the deontological (or duty-generating) approach, intrinsic value implies a set of rights that include a right of existence. Under this approach, something with intrinsic value is irreplaceable, implying that a loss cannot be offset or "compensated" by having more of

[3] This assumption rules out fixed proportions preferences between the different categories of values.

something else. For example, a human person's own life is of intrinsic value to that person because it cannot be offset or compensated by that person having more of something else. This approach has its roots in the writings of the philosopher Immanual Kant, who wrote extensively about intrinsic value (e.g., Kant translated in 1987). However, Kant used the concept of rationality to determine the realm of beings that have intrinsic value and rights. He argued that human beings were the only beings who were rational and thus that only human beings have intrinsic value and rights. In this sense, Kant's views were strictly anthropocentric. Since Kant's writings, others have suggested alternative criteria for determining the realm for intrinsic value and rights (see footnote 31 in Callicott, 2004) and hence have argued that rights should extend to nonhumans, including animals (either individual animals or species) and in some cases all biological creatures (i.e., all plant and animal life) or the biota collectively. The modern notion of intrinsic value (as used in the context of ecosystem valuation) reflects the notion that rights should be extended beyond human beings (Stone, 1974).[4]

As discussed in more detail below, the economic approach to valuation is an anthropocentric approach based on utilitarian principles. It includes consideration of all instrumental values, including existence value. Environmental policy and law may also be based on intrinsic value, as exemplified by the Endangered Species Act of 1973. Because it is utilitarian based, economic valuation assumes that the potential for substitutability between the different sources of value that contribute to human welfare. The main categories of value that are not captured by the economic approach are non-anthropocentric values (e.g., biocentric values) and intrinsic values on which the concept of rights is based.

Finally, it is important to keep in mind that economic valuation is based on the notion that the values assigned by an individual reflect that individual's preferences or marginal willingness to trade one good or service for another, and that societal values are the aggregation of individual values. At any point in time, individual preferences can be influenced by a variety of factors, including culture and information, which can change over time. In addition, an individual's willingness to trade one good for another will reflect the amounts of the goods and services currently available to him, which will in turn depend at least partially on income. If income changes over time, the economic measure of value for an individual can be expected to change as well. For these reasons, the values measured through economic valuation are inherently time- and context-specific.

## Quantifying Values

Recognition that ecosystems or ecosystem services are valuable, possibly in a variety of ways or for a variety of reasons, does not necessarily imply a *quantification* of that value (i.e., its *valuation*).[5] In fact, those people who affirm the intrinsic value of ecosystems object to the very idea of trying to quantify the value of environmental goods and services (see, for example, Dreyfus, 1982; MacLean, 1986; Sagoff, 1993, 1994, 1997). For them, that would be as objectionable as quantifying the value of human life. The quantification of the value of ecosystems is by definition anthropocentric since humans are doing it. In addition, it implies a

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[4] A good reference regarding the relationship between intrinsic value and legal rights is Christopher Stone's *Should Trees Have Standing? Towards a Theory of Legal Rights for Natural Objects* (Stone, 1974).

[5] It is important to distinguish between "values," which are an attribute of a good or service, and "valuation," which is the process of quantifying that attribute.

ranking of values (i.e., a statement of which goods or services are "more valuable," and possibly by how much). Some people object to one or both of these implications of quantification as being analogous to ranking the value of different human beings based, for example, on gender or ethnicity.

However, there are a number of contexts in which quantification of such values may be useful or even necessary, including (1) informing policy decisions in which trade-offs are considered, (2) providing damage estimates for natural resource damage assessment (NRDA) or similar cases, and (3) incorporating environmental assets and services into national income accounts.[6] For example, if an environmental policy decision involves a trade-off in the choice between providing one ecosystem service (such as a particular habitat or an ecological service) and providing another good or service (such as agricultural output), then information about the relative values of these alternative goods or services can lead to better-informed and more defensible choices. This requires a ranking of values, which follows from quantification. A recognition that quantification or valuation may be useful or necessary in informing policy decisions is explicit in the remainder of the committee's statement of task (see Box ES-1). Given the committee's charge, the remainder of this report focuses on the role of valuation in the context of policy decisions and improved environmental decision-making. Although not the focus of this study, the committee believes that quantification is also important (in fact, necessary) in the other two contexts as well. In NRDA cases, a quantification of lost value is necessary to determine the compensation that must be paid by responsible parties.[7] Similarly, in order to incorporate changes in environmental and other natural assets into national income accounts, these changes must be quantified in a manner comparable to the quantification of the other components of national income (Heal and Kriström, 2003; NRC, 1999).

If quantification is deemed to be a useful or necessary input for policy decisions, a particular quantification or valuation approach must be selected. As noted above, given the committee's charge, this report focuses on the quantification embodied in the economic approach to valuation. In this approach to valuation, the metric that is used to quantify values in nearly all applications is a monetary metric, such as U.S. dollars.[8] In the context of ecosystem goods or services that are bought and sold in markets, dollars or some other currency provide a natural metric for quantification since such prices, absent any market distortions, reflect the consumer valuation of that good (see further discussion in Chapter 4). Thus, when policies involve trade-offs between market goods (already valued in dollar terms) and ecosystem services that are not traded in markets, quantifying the value of these nonmarket services using the same metric (e.g., a dollar metric) allows a direct assessment of the trade-offs.

However, the use of a dollar metric for quantifying values is based on the assumption that individuals are willing to trade the good being valued for something else that can also be quantified by the dollar metric. It thus assumes that the good being valued is in principle substitutable or replaceable with other goods or services that are also of value and that money

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[6] Note that the type of quantification that is necessary can vary across these different contexts. For example, NRDA requires a point estimate of the total damages or lost benefits from an environmental reduction in ecosystem services resulting from some event (e.g., an oil spill). In contrast, in a policy context, quantification of the value of a subset of services may be sufficient (see Chapters 5 and 6 for further discussion).

[7] Quantification of values is not necessary if compensation is measured in physical units (e.g., when based on habitat equivalency). However, a habitat equivalency approach to compensation implicitly assumes that the value of the restored or replaced habitat is equivalent to the value of the degraded one.

[8] Some have advocated the use of energy analysis as an alternative currency or metric for measuring value. See Chapter 3 and Box 3-7 for further information.

can buy; this reflects the utilitarian principles that underlie economic valuation.[9]

### The Role of Valuation in the Policy Process

Although economic valuation requires a quantification of values, the specific design of the valuation exercise should depend on its purpose or the role that it will play in the policy process. One approach is to base policy decisions regarding preservation of environmental resources on moral principles, stemming from a political consensus about what is morally right or wrong. While adherence to moral principles relating to intrinsic value will inevitably involve trade-offs, under this approach these trade-offs are of little or no consequence to the policy choice. If policy choices are to be based on the notion of intrinsic values and rights, then these rights have to be identified, but the values are implied by that identification need not be quantified in order to choose among alternatives (unless the decision to protect one intrinsic value implies a loss of something else with intrinsic value). Thus, with this decision rule, valuation of ecosystem services has no effect on policy choices and hence plays a very limited role (see Goulder and Kennedy, 1997).[10]

Strict utilitarianism, on the other hand, implies that a decision is based solely on economic efficiency, that is, maximization of the net benefits to society (Goulder and Kennedy, 1997). This decision rule is implemented through the use of benefit-cost analysis (BCA). Economic valuation plays a central role in the application of BCA, since BCA requires an estimate of the benefits and costs of each alternative using a common method (economic valuation) and metric (dollars) so that the two can be compared. The comparison of costs and benefits allows an explicit consideration of the trade-offs that are inevitably involved in most environmental policy decisions. It recognizes that achieving a particular objective or goal such as preservation of a particular ecosystem comes at a cost, since the resources that must be devoted to this preservation are not available for use in providing other goods and services. A typical BCA asks whether the benefits of that preservation are "worth" the costs involved. In this sense, it ensures that the limited resources used to provide goods and services to society are used in the most efficient way—that is, to achieve the greatest net benefit.

In addition, a benefit-cost approach provides a means of combining heterogeneous views of what is desirable. Although some may prefer preservation of the environment or a particular ecosystem, others may prefer an alternative (e.g., development of the land). These different views can stem from differences in an individual's net benefits from the alternatives. Those who realize a net gain from preservation would be expected to prefer preservation, whereas those who realize a net gain from the alternative are likely to prefer it. The benefit-cost approach provides a mechanism for combining these disparate views to reach a decision that incorporates both

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[9] Several environmental philosophers argue that while a monetary metric is an appropriate metric for utilitarian values, it is inappropriate for non-utilitarian values such as non-anthropocentric intrinsic values or values based on notions of morals, rights, and duties (deontological values) (e.g., Sagoff, undated and 1997; Callicott, 2004). This raises the question of what, if any, metric might be used to quantify, or at least rank, these non-utilitiarian values. Callicott (2004) suggests use of a "penalty metric." He argues that the severity of the penalties imposed for violations of certain types of protections that reflect intrinsic value provides a democratically determined measure, or at least ordinal ranking, of those values.

[10] Of course, valuation could be used in this context to determine whether adherence to a moral principle came at a net cost or benefit to society. However, under such an approach, this information would be a "curiosity" rather than a determinant of the policy choice.

perspectives. Of course, in doing so, it assigns equal weights to the net benefits of all individuals, a property of BCA that may draw criticism (Azar, 1999; Layard, 1999; Potts, 1999).

If BCA is to be used to evaluate environmental policy options, it is imperative that all costs and benefits be considered.[11] In particular, for policy decisions that impact ecosystems, the benefits that the ecosystem generates through the various goods and services it provides must be included in calculating the benefits of preserving the ecosystem or the costs (forgone benefits) of allowing it to be degraded. As noted in Chapter 1, failure to assign a dollar value to these benefits (e.g., on the principle that they cannot be valued accurately or that the values are "incalculable") effectively assigns them a zero value or a zero weight in the calculation of net benefits, implying that changes in those services will not be incorporated into the net benefit calculation (Epstein, 2003).

Political and legal decisions are often made on the basis of information about many sources of value, including intrinsic and moral values, as well as economic values, and some decision rules seek to incorporate different types of values explicitly. For example, decision rules that imply adherence to moral principles or a premise of intrinsic value unless the cost is too high (as under a "safe minimum standard" rule; see Chapter 6 for further information) incorporate concern about both intrinsic value and economic welfare, and implicitly allow some trade-offs between the two. Similar trade-offs are also implied by decision rules that apply a benefit-cost test to environmental policy choices but constrain the decisions to ensure that certain conditions reflecting intrinsic value are not violated. Possible constraints include ensuring (1) that basic notions of justice and fairness are not violated, (2) that populations or levels of critical ecosystem services do not fall below standards necessary to ensure their continuation, and (3) that uncertainties regarding outcomes are not deemed too great. In such cases, information about benefits and costs as determined by economic valuation will be a useful input into the policy decision but will not solely determine it, since the net benefits from the various alternatives will be only one of the factors considered when making a policy choice.

Examples of different weights put on intrinsic values versus utilitarian welfare can be found throughout environmental policies in the United States. For example, the Clean Air Act requires a periodic assessment of the costs and benefits of the act, although it clearly states that the costs or impacts of any standard or regulation promulgated under the act shall not be a basis for changes that preclude the U.S. Environmental Protection Agency (EPA) from carrying out its central mission to "protect human health and welfare." Thus, information about costs and benefits is intended to inform but not drive policy decisions. In contrast, Executive Order 12291[12] required a strict cost-benefit approach to evaluating regulations. The order stated that "regulatory action shall not be undertaken unless the potential benefits to society for the regulation outweigh the potential costs to society." This order, and a related order (Executive Order 12866), were later replaced by Executive Order 13258, issued in 1996, which replaced the strict benefit-cost criterion for decision-making with a weaker version that instead simply required that the benefits of the regulation *justify* the costs (OMB, 1996; see also Chapter 4). Under this more recent order, BCA is an input into regulatory decisions but not the sole criterion for them.

[11] In some cases, the decision implied by a benefit-cost analysis may be clear without a full quantification of all values. For example, if a proposal or project would pass a benefit-cost test with a complete quantification of costs and an incomplete quantification of benefits, then it would also pass with a complete quantification of benefits. In such a case, quantification of the remaining benefits would not change the results of the test.

[12] Executive Order 12291. February 19, 1981. Federal Register 46(33).

Other environmental policies appear to reject more explicitly a consideration of benefits and costs in favor of an approach based on intrinsic value and rights. For example, Callicott (2004) has argued that the protection granted to species under the Endangered Species Act (ESA) is based primarily on principles regarding the duty to preserve species because of their intrinsic value. In *Tennessee Valley Authority vs. Hill*, the U.S. Supreme Court found that although "the burden on the public through the loss of millions of unrecoverable dollars would [seem to] greatly outweigh the loss of the snail darter. . . , *neither the Endangered Species Act nor Article III of the Constitution provides federal courts with authority to make such fine utilitarian calculations*" [emphasis added]. On the contrary, the plain language of the act, buttressed by its legislative history, shows clearly that Congress viewed the value of endangered species as "incalculable" (e.g., Telico dam-snail darter case; U.S. Supreme Court, 1978).[13] In response to this finding, Congress immediately amended the ESA to allow at least the possibility of consideration of benefits and costs and to create a committee with authority to grant exceptions to the law's prohibitions under very limited conditions that consider, but do not simply compare, benefits and costs.

It is clear from the preceding overview that in many policy contexts relating to the use and preservation of environmental resources, some consideration is given to the magnitude of benefits and costs, even though this information is likely to be only one of many possible considerations that influence policy choice. To provide this information, those benefits and costs must be measured, and economic valuation provides a means of measuring them. It is the judgment of this committee that having the best available and most reliable information about the economic valuation of ecosystem services will lead to improved environmental decision-making. It will allow policymakers to identify and evaluate trade-offs and, if appropriate, incorporate a consideration of the trade-offs into environmental policy design.

## Framing the Valuation Question

In order to be useful in the evaluation of environmental policy options, the valuation exercise should be designed or framed to provide the necessary information to policymakers. A number of dimensions are important in framing the analysis. Some of these dimensions are discussed briefly below (see also Chapter 6).

First, it is important to recognize that policy choices, and the benefits and costs associated with them, imply *changes* in environmental quality or the level of environmental services (e.g., changes in ecosystem goods and services), either positive or negative, and that the valuation exercise is the quantification of the value of those changes.[14] Thus, in a policy context, economic valuation is not concerned with quantifying the value of an entire ecosystem (unless the policy under consideration would effectively destroy the entire ecosystem); rather, it is concerned with translating the physical changes in the ecosystem and the resulting change in ecosystem services into a common metric of associated changes in the welfare (utility or "happiness") of members of the relevant population. Thus, the valuation of ecosystem services should be framed in terms of valuing the changes in those services implied by different policy

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[13] See Erdheim (1981) for a discussion of this seminal case.

[14] An important consideration is the benchmark used for measuring these changes. Different benchmarks imply different assumptions about property rights and require different valuation measures. The link between valuation measures and property rights is discussed later in this chapter.

choices.

A second important dimension of framing is the scope of the analysis. Scope refers to the inclusion or exclusion, by choice or necessity, of certain ecosystem functions or services and/or certain types of value. Thus, a valuation exercise may focus on only a subset of ecosystem services; for example, an exercise might seek to value changes in flood control or water purification services but not changes in the quantity or quality of habitat. Similarly, the valuation exercise may focus (by necessity) on the quantification of certain types or sources of value and may not capture other sources. Although a broader scope provides a more accurate picture of the total impact of the policy change, in some policy contexts a partial approach may be sufficient. For example, if the results of a benefit-cost analysis based on a measure of the partial value of ecosystem preservation imply that the benefits of a particular policy or activity outweigh the costs, then inclusion of additional benefits (by valuing additional services or including additional sources of value) will only reinforce this conclusion (see also footnote 11).

The outcome of the valuation exercise will also depend on its spatial or geographic scale (see Chapters 3 and 5 for further information). Spatial scale has two components. The first is definition of the geographic extent of the relevant ecosystem(s). In defining the physical impacts of a given policy, one can restrict consideration to fairly localized impacts or consider spillover impacts on related ecosystems that are not impacted directly but change indirectly through those linkages.[15] Consideration of these indirect impacts will yield a more inclusive analysis, but these indirect effects may be difficult to identify and quantify accurately. In addition, some policies (particularly at the national level) can affect many ecosystems. For example, a categorical exclusion under the National Environmental Policy Act (NEPA) of federal activity in all wetlands 10 acres or less in size will affect the hundreds or thousands of wetlands across the United States. In such cases, the aggregate impact across all affected ecosystems should be valued.

The second component of spatial or geographic scale is definition of the relevant population (i.e., the stakeholders). In estimating the value that individuals place on ecosystem changes, one must identify which individuals (whose values) to include. In other words, what is the relevant population for estimating the benefits and costs of the policy change? For example, in valuing possible damages from a major oil spill, should calculations reflect damages to the local population, to the population within the state, to the population within the nation, or to the world population? Because an oil spill that leads to loss of wildlife may negatively impact those outside the local area who value the existence of the animals, the aggregate measure of damages will generally vary directly with the extent of the population considered (Carson et al., 2001). The appropriate population to include will depend on the perspective of the decision-maker, his or her jurisdiction, and the target population of concern to the decision-maker when assessing the aggregate welfare impacts of the policy change. Thus, local officials may be concerned primarily with the costs and benefits borne by their local constituents, while national policymakers can be expected to take a broader view.

In addition to the spatial or geographical scale, the valuation exercise is also affected by the temporal scale of the analysis (i.e., the period of time over which benefits and costs are distributed). Most policy impacts last for extended periods, and some last (effectively) forever because they lead to irreversible changes. This is particularly likely in the context of ecosystems, where stock effects are important and losses of key ecosystem services may be

[15] This distinction is comparable to the economic distinction between partial and general equilibrium analysis (see further discussion below).

irreversible. When the benefits and/or costs extend over time, the period of analysis becomes a key factor in determining the results of a valuation exercise. For example, if land conversion for development purposes causes irreversible loss of critical habitat, an analysis that considers only a short time period will not accurately assess the benefits and costs of that conversion. In addition, the analysis should account for differences in the timing of impacts across alternatives. One approach to this is the use of discounting to weight impacts differently depending on when they occur. The meaning and use of discounting are discussed later in this chapter (see also Chapter 6). At this point, it is sufficient to note that the temporal framing of the valuation exercise—the time period chosen and the method used to reflect differences in the timing of impacts—plays a crucial role in determining its results.

The discussion thus far suggests that the quantification of ecosystem value using the economic approach to valuation can and does play an important role in environmental policy analysis and decision-making. However, the results that emerge from this quantification or the valuation exercise will be influenced significantly by the way in which the valuation question is framed. To provide meaningful input to decision-makers, it is imperative that the valuation exercise seeks to value the *changes* in ecosystem goods or services attributable to the policy change, that the scope considers all relevant impacts and stakeholders, and that the temporal scale of the analysis is consistent with the scale of the impacts. The results will also depend on a number of methodological and data issues. These issues are discussed in detail in Chapter 4 and illustrated through the case studies provided in Chapter 5.

## THE ECONOMIC APPROACH TO VALUATION

Having discussed economic valuation and its role in general terms, a more detailed discussion of the economic approach to valuation follows. As noted earlier, the economic concept of value is based on an anthropocentric, utilitarian approach to defining value based on individual preferences. As such, it does not encompass all possible sources of value. However, it is much broader than the narrow concept of commercial or financial value, and includes all values, tangible as well as intangible, that contribute to human satisfaction or welfare. This broad definition is reflected in the "total economic value" framework that underlies economic valuation and is described below.

### The Total Economic Value Framework: Use and Nonuse Values

The *total economic value* (TEV) framework is based on the presumption that individuals can hold multiple values for ecosystems. It provides a basis for a taxonomy of these various values or benefits. Although any taxonomy of such values is somewhat arbitrary and may differ from one use to another, the TEV framework is necessary to ensure that all components of value are given recognition in empirical analyses and that "double counting" of values does not occur when multiple valuation methods are employed (Bishop et al., 1987; Randall, 1991). It is important to state that the TEV framework does not imply that the "total value" of an ecosystem should be estimated for each policy of concern. Even a marginal change in ecosystem services can give rise to changes in multiple values that can be held by the same individual, and the TEV framework simply implies that all values that an individual holds for a change should be counted.

In the simplest form, TEV distinguishes between *use* values and *nonuse* values. The former refer to those values associated with current or future (potential) use of an environmental resource by an individual, while nonuse values arise from the continued existence of the resource and are unrelated to use. Typically, use values involve some human "interaction" with the resource whereas nonuse values do not. The distinction between use and nonuse values is similar but not identical to the distinction between instrumental and intrinsic value discussed earlier. Clearly, use values are instrumental and utilitarian, but, as noted above, the concept of existence value is not identical to the notion of intrinsic value, because the latter is deontological and includes non-anthropocentric values while the former does not.

Within the TEV framework an individual can hold both use and nonuse values for the services of an aquatic ecosystem. Consider an oil spill on a popular coastal beach resulting in forgone recreational trips to the beach—this is a lost use value. In addition, the oil spill could damage the ecosystem in ways that would not affect beach use and that beach users would never observe. It might, for example, kill marine mammals that live off the beach and are not seen by beach users, and beach users, as well as those who do not visit the beach, might experience a loss because of this ecosystem damage. The loss by those who do not visit the beach would be a loss of nonuse value, though there could also be a loss of nonuse value on the part of beach users. The TEV framework implies that analysts proceed to investigate the potential loss in use and in nonuse values of beach users and in nonuse values of people who do not visit the beach. It is not necessary to estimate the total value of the coastal ecosystem, only the total loss in value associated with the oil spill.

A number of TEV frameworks have been proposed in recent decades (e.g., Bishop et al., 1987; Freeman, 1993a; Randall, 1991). Although varied in detail and application, the distinction between use and nonuse values is a fundamental theme. The TEV framework, as applied to typical aquatic system services for the purposes of this report, is illustrated in Table 2-1. In the discussion below, distinctions are drawn between the components of TEV, but when people hold both use and nonuse values, the literature cited above argues for estimating peoples' TEV rather than estimating the components and then adding the component estimates to compute a TEV. However, the discussion of valuation methods in Chapter 4 shows that some methods are better able to measure selected components of TEV than others.

TABLE 2-1 Classification and Examples of Total Economic Values for Aquatic Ecosystem Services

| Use Values | | Nonuse Values |
|---|---|---|
| Direct | Indirect | Existence and Bequest Values |
| Commercial and recreational fishing | Nutrient retention and cycling | Cultural heritage |
| Aquaculture | Flood control | Resources for future generations |
| Transportation | Storm protection | Existence of charismatic species |
| Wild resources | Habitat function | Existence of wild places |
| Potable water | Shoreline and river bank stabilization | |
| Recreation | | |
| Genetic material | | |
| Scientific and educational opportunities | | |

SOURCE: Adapted from Barbier (1994) and Barbier et al. (1997).

### *Use Values*

Use values are generally grouped according to whether they are *direct* or *indirect*. The former refers to both *consumptive* and *nonconsumptive* uses that involve some form of direct physical interaction with the resources and services of the system. Consumptive uses involve extracting a component of the ecosystem for an anthropocentric purpose such as harvesting fish and wild resources. In contrast, nonconsumptive direct uses involve services provided directly by aquatic ecosystems without extraction, such as use of water for transportation and recreational activities such as swimming. Although nonconsumptive uses do not involve extraction and hence diminution in the quantity of the resource available, they can diminish the quality of aquatic ecosystems through pollution and other external effects.

It is also increasingly recognized that the livelihoods of populations in areas near aquatic ecosystems may be affected by certain key *regulatory ecological functions* (e.g., storm or flood protection, water purification, habitat functions) (Daily, 1997). The values derived from these services are considered indirect, since they are derived from the support and protection of activities that have directly measurable values (e.g., property and land values, drinking supplies, commercial fishing). For example, mangrove swamps may provide a "storm protection" function in that they may stop coastal storms from wreaking havoc on valuable coastal properties and infrastructure (Janssen and Padilla, 1999). Activities such as reading a book or magazine article about ecosystems, or watching a nature program, are also thought to provide indirect use values.

### *Nonuse Values*

Many natural environments are thought to have substantial existence values; individuals do not make use of these environments but nevertheless wish to see them preserved "in their own right" (Bishop and Welsh, 1992; Boyle and Bishop, 1987; Freeman, 1993b; Madariaga and McConnell, 1987; Randall, 1991; Smith, 1987). The terms "existence," "nonuse," and "passive" use are generally used synonymously in the literature. For the purposes of this report, nonuse values refer to all values people hold that are not associated with the use of an ecosystem good or service. Use values typically arise from a good or service provided by ecosystems that people find desirable. Nonuse values need not arise from a service provided by an aquatic ecosystem; rather, people may benefit from the knowledge that an ecosystem simply exists unfettered by human activity (e.g., Crater Lake). The latter is what was traditionally known as a "pure" existence value in the literature. Other motivations for nonuse values are bequest and cultural or heritage values. The empirical literature generally does not attempt to measure values for individual aspects of nonuse values, but focuses on the estimation of nonuse values irrespective of the underlying motivations people have for holding this value component.

The economic valuation of the impacts of the *Exxon Valdez* oil spill on the aquatic and related ecosystems of Prince William Sound, Alaska, highlights the importance of nonuse values in natural resource damage assessments and project appraisals (Carson et al., 1992). The *Exxon Valdez* study revealed that many Americans who have not visited Alaska and never intend to do so nevertheless place high values on maintaining the pristine and unique but fragile coastal and aquatic ecosystems of Alaska. In the context of the *Exxon Valdez* study, questions were raised about the accuracy with which nonuse values can be estimated (Hausman, 1993; NOAA, 1993). This issue is discussed in greater detail in Chapter 4.

### Measurement Using a Monetary Metric: WTP Versus WTA

Economic valuation is concerned with how to estimate the impact of changes in ecosystem services on the welfare of individuals and is based on the principles of utilitarianism. If ecosystem changes result in individuals' judging that they are worse off, one would like to have some measure of the loss of welfare to these individuals. Alternatively, if the changes make people better off, one would want to estimate the resulting welfare gain.

The basic concept used by economists to measure such welfare gains and losses is rooted in the utilitarian notion that for any individual, the different sources of value that affect the individual's utility are potentially substitutable; that is, the individual is willing to trade a reduction in one source of value for an increase in another in a manner that leaves his or her overall utility unchanged. The essence of this approach is to value a change by determining what people would be willing to trade (i.e., to receive or to give up) so they would be equally satisfied or happy with or without the change.

Consider, for example, a case in which a freshwater lake can be restored to enhance sportfishing opportunities. An economic measure of the benefit of such an improvement to recreational anglers is the maximum that anglers would be willing to pay for this improvement in fishing if he or she had to pay. Each angler's maximum willingness to pay should represent how much money the angler is prepared to give up in exchange for the increase in individual enjoyment gained from the improved recreational fishing. It represents the reduction in income that would be necessary to offset exactly the gain in angler utility resulting from the restoration, thereby leaving anglers at the same utility level as they were prior to any restoration. Maximum willingness to pay could then be aggregated for all anglers who benefit to determine the total benefits of the project.[16] This aggregation, in turn, would facilitate an assessment of whether public funds should be spent on the project.

An alternative measure of the value of the improvement in recreational fishing from restoration of the lake is based not on anglers' willingness to *pay* for the improvement but rather on the amount they would be willing to *accept* to forgo the improvement. If the improvement is promised, then failure to provide this improvement (i.e., failure to restore the lake) would reduce the utility of anglers relative to the level they would have attained with the restoration. The value of this loss or the forgone benefit from restoration can be measured by the minimum amount of income that the anglers would be willing to accept as compensation for forgoing that benefit. The increase in income (i.e., the compensation) would have to increase the utility of anglers by exactly the same amount as the reduction in utility stemming from the failure to restore the lake, so that the combined effect would be to leave utility unchanged (i.e., leave the anglers just as well off without the restoration as they would have been with it).

The preceding example illustrates the two alternative measures of value that are used in economic valuation: WTP and WTA. Each measure looks at potential trade-offs between money and the good or service being valued that leave utility unchanged from some base level. They differ, however, in the base level of utility that is maintained when the hypothetical trade-

[16] It is important to note that the concept of willingness to pay does not rely on the individual's actually paying for the change.

off is made. In valuing an improvement in environmental quality or services, WTP considers trade-offs that would leave utility at the level that existed prior to the improvement (the pre-change utility level), whereas WTA considers the utility level that would exist after the improvement (the post-change utility level).

In some cases such as when valuing small price changes, WTP and WTA measures of value can be expected to be quite close, differing only because of the different income levels implied by paying rather than receiving compensation (Willig, 1976). However, for many environmental goods and services, the two can be substantially different. In particular, Hanemann (1991) has shown that when valuing changes in the quantities of goods or services available for which there are no close substitutes (including many ecosystem services), the two measures of value can yield quite different results. For environmental improvements, the amount an individual is willing to accept to forgo that improvement will normally be greater than the amount he or she would be willing to pay to ensure it (WTA > WTP).

Because WTP and WTA measures of ecosystem services could differ significantly, a key issue in the use of economic valuation in this context is the choice between these two possible measures of value. As noted above, the conceptual difference lies in the base level of utility that each is designed to ensure. This reflects a difference in the assumption regarding the underlying allocation of property rights or, equivalently, the baseline levels of utility that society collectively agrees to ensure to each individual within that society. Consider again the case of lake restoration. If anglers do not have a right to the improved conditions, then society is not collectively prepared to ensure them a level of utility that includes the restoration. If these anglers want restoration, then in theory they would have to "buy" it from the rest of society. In such a case, WTP is the appropriate economic measure of the value of the improvement. Conversely, if anglers have a right to the improved conditions, then if society wants to use the resources for other purposes, in theory it would have to buy the right to do so from the anglers and pay or otherwise compensate them for failure to restore the lake. In such a case, WTA is the appropriate economic measure of the value of the water quality improvement.

Economic theory, and hence economic valuation, provides no basis for choosing between the alternative property rights regimes and therefore no basis for preferring one measure of value over the other. Property rights are determined collectively by society. In addition, virtually all theories of property rights recognize that they are not absolute or strong but represent only "weak" rights, insofar as they are subject to modification and based on community welfare in ways that strong rights (e.g., a right to life) are not. They are weak rather than strong because they are not considered essential to human dignity in the way that rights to life or to equal protection are (Dworkin, 1977).

Although in theory economic valuation can seek to measure either WTP or WTA depending on the underlying assignment of property rights, it is common to use WTP as an empirically reliable measure. The primary reason is that most of the existing economic methods for estimating values capture WTP but not WTA (see Chapter 4 for further information). The use of WTP may be inappropriate in a given case because of the implicit property rights assumption embedded in it. However, even in cases where WTA would be the appropriate measure, WTP may still be a reasonable proxy for WTA. In theory and practice, the absolute value of willingness to accept usually exceeds the absolute value of willingness to pay (Hanemann, 1991; Horowitz and McConnell, 2002). Thus, WTP can be viewed as a lower bound for WTA and hence as a lower-bound for the value of the improvement. In some contexts, a lower bound estimate of values will be sufficient to inform policy decisions. For example, if the benefits of an increase in ecosystem services exceed the costs when those benefits are measured using WTP, they would also have exceeded costs if measured using a higher WTA. However, if a WTP measure of benefits was lower than cost in a context in which WTA was the correct measure to use, then it is still possible that benefits would have exceeded costs had WTA been used.

In addition to the difference regarding the implicit assumption with respect to underlying property rights, WTP and WTA also differ in another important aspect, namely, the role of income limitations. Clearly, the amount that an individual is willing to pay for an environmental improvement depends on the amount that he or she is *able* to pay. In other words, WTP is constrained by an individual's income since he or she could never be willing to pay more than the amount available. WTA, on the other hand, is not income constrained. The amount of compensation that would be required to compensate an individual for accepting a lower level of environmental quality can exceed a person's income. This difference has important implications in measures of aggregate net benefits. Income constraints imply that, all else being equal, low-income individuals will have a lower WTP than wealthier individuals simply because of their lower ability to pay. This implies that the preferences of wealthy people will get more weight than those of poorer people in net benefit calculations based on WTP. This feature of WTP should be borne in mind when using this measure of value.

## Uncertainty and Valuation

Estimates of the values of ecosystem services are frequently somewhat uncertain for a variety of reasons. Chapter 6 explores the major sources and types of uncertainty, indicates which are most significant, and discusses their consequences in ecosystem services valuation. This discussion includes the problems posed by uncertainties about models and parameters, and how analysts and decision-makers can and should respond. Sensitivity analysis and Monte Carlo simulation are discussed as a possible analyst response to model and parameter uncertainties, while risk aversion, quasi option values, adaptive management, safe minimum standards, and the precautionary principle are discussed in the context of use by decision-makers.

## Discounting: Utility versus Consumption

In many ecosystem valuation contexts, the impacts of a particular policy choice will extend over time, and hence an attempt must be made to estimate the costs and benefits not only for current years but well into the future. Deriving an aggregate measure of costs or benefits that reflects their change over time requires an aggregation method that appropriately incorporates the timing of benefits and costs. The most commonly used approach in economic valuation is discounting, that is, weighting future costs and benefits differently than current costs and benefits when summing over time.

The desirability of discounting future costs and benefits has been the subject of intense debate (Heal, 1998; Portney and Weyant, 1999). The simplest explanation of discounting can be found in the financial context. People generally agree, for example, that accountants are correct to discount future income. If a person will receive an income of $20,000 a year for the next 30 years, most people would agree that it is unreasonable to value that total income at 30 times

\$20,000. Instead, a more reasonable valuation would be \$20,000 for the first year, plus \$20,000 discounted by some rate (such as 5 percent) for the second year, plus the amount from the second year, discounted by an additional 5 percent, for the third year, and so on. The rationale for such discounting is the productive power of the economy that converts commodities at one time into a greater quantity of commodities at a later time. If one ignores inflation, then money represents a quantity of purchasing power over economic commodities, and therefore commodities available at an earlier time are worth more than commodities available only at a later time. If the economy remains productive, then (even on a simple level) it is easy to see that money at a later time is worth less than money at the present time because, for example, money this year can be converted into more money in the future by depositing it into a bank to earn interest.

However, the issues raised by the use of discounting in cost-benefit analysis, project evaluation, and ecosystem valuation go far beyond the simple arithmetic of compound interest on bank balances. It is important to realize that there are two different types of discounting may be practiced—utility discounting and consumption discounting. This distinction is absolutely central, although unfortunately it is not as widely understood. The properties of and justifications for these two rates are quite different, and some of the arguments that apply to one are not relevant in the context of the other (Heal, 2004).

This chapter provides only a brief summary of the underlying issues, which are quite complex and the subject of a massive literature.[17] What is normally referred to as "the discount rate" is in fact the *utility discount rate*, also known as the pure rate of time preference, the social rate of discount, or the social rate of time preference.[18] This is the rate to which Frank Ramsey's famous strictures apply and indeed those of Roy Harrod as well.[19] There is no compelling reason for this discount rate to be positive; the value of the utility discount rate reflects the relative valuations that are placed on present and future generations. If one is convinced that future generations should be valued less than present generations, then a positive utility discount rate should be chosen; otherwise this rate should be zero.

The *consumption discount rate* is conceptually and operationally different from the utility discount rate. The utility discount rate, as emphasized above, is intended to represent the relative weights put on present and future utilities. It expresses society's preferences for distribution between generations, with a zero rate representing equal weights for all generations, and a positive rate implying less weight to future people. In contrast, the consumption discount rate represents the weights placed on increments of consumption at different dates. It answers the question, How does one value an extra dollar's worth of consumption (instead of an extra unit of utility) today relative to an extra dollar's worth of consumption in the future?

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[17] For a more detailed discussion, see Heal (2004).

[18] This is the rate $r$ in the utilitarian maximand $\int_0^\infty U(c)e^{-rt}dt$. In the utilitarian approach a proposed policy is evaluated by the weighted sum of the utilities accruing at different points in time. The weight placed on utility at time $t$ is given by $e^{-rt}$, an exponential function of time. The utility discount rate is the rate at which this weight—the weight placed on utility at time $t$—decreases with time. It is the proportional rate of change of $e^{-rt}$ with $t$, which is of course just $r$. The reason for calling this the utility discount rate is obvious; it is the rate at which one discounts utility.

[19] Frank Ramsey was an influential economist and mathematician at Cambridge, United Kingdom, in the 1920s. He remarked that "discounting is ethically indefensible and arises purely from a weakness of the imagination" (Ramsey, 1928). Roy Harrod, an Oxford University economist of the same generation, wrote similarly that discounting is a "polite expression for rapacity and the conquest of reason by passion" (Harrod, 1948).

Even if future utilities are valued the same as present utilities (i.e., there is a zero utility discount rate), one may still value an increment of consumption 20 years in the future differently from the same increment today. There are several reasons for this. One reflects changes in wealth or the standard of living over time. Suppose, for example, that people 20 years from now are expected to be wealthier than those today. If the extra utility generated by additional consumption diminishes with income, then providing the additional consumption in the future when people are wealthier will yield less of an increase in utility than providing the same additional consumption today. This suggests that future consumption should be discounted. If this were done, however, it would not reflect a judgment about the relative merits of present and future people, which is what the utility discount rate does. Rather, it would reflect a distributional judgment about the relative merits of extra consumption going to richer or poorer people, quite independent of the dates at which they live. If this approach is accepted, it implies a positive consumption discount rate when living standards are rising over time and, conversely, a negative rate when they are falling.

The distinction between utility and consumption discounting is important in the context of environmental issues (Heal, 2004). One might feel that access to aquatic ecosystem services will decrease over time as a result of human pressures on natural habitat, and that, consequently, peoples' marginal valuations of these services will increase as they become scarcer. As a result, the value of incremental ecosystem services will rise over time and the consumption discount rate to be applied to these will be negative rather than positive. That is to say, increments in the future will be worth more than those in the present—not because they are in the future but rather because they are being made available at a later date when they are scarcer. This reflects diminishing marginal utility or valuation rather than the result of futurity.

It follows from this discussion that the consumption discount rate is quite flexible and reflects many different characteristics of the underlying problem. If people are concerned with ecosystem goods and services, which are expected to be scarcer in the future than in the present, then the consumption discount rate may be negative, meaning that a unit of consumption in the future would be valued more than a unit at present. If income levels are rising over time, then future income levels will be higher than those at present, so the marginal valuation of income will fall over time and the consumption discount rate will be positive (i.e., the future should be discounted).

The preceding discussion highlights the existence of two quite distinct concepts of discounting—utility and consumption discounting. It argues that there is no compelling argument for discounting utility, but that there may be reasons for discounting consumption, although the appropriate rate may be positive or negative. When is it appropriate to use the consumption discount rate in ecosystem valuation and when should the utility discount rate be used instead?

In general, the utility discount rate should be used when the policy under consideration is such as to lead to changes in the overall utility or welfare levels of the economy, or at least a significant subsector of it. In economic terms, the utility discount rate is applicable in the context of general equilibrium analyses. The consumption discount rate, on the other hand, is applicable in the context of partial equilibrium problems. These are problems in which only a small part of the economy is being affected by our decisions, and these decisions have only a small impact on overall consumer welfare. Because all of the environmental valuation problems considered in this report are of a partial equilibrium nature, the relevant discount rate to be considered is the consumption rate, which may have either sign. The committee emphasizes that the consumption discount rate is the rate of change of the value placed on an increment of consumption as its date changes. It is not a number that the analyst chooses a priori but one that emerges from the characteristics of the economy, such as whether consumption of the ecosystem good at issue increases or decreases over time. Given this interpretation, one does not argue about whether to discount consumption or at what rate. Discounting consumption—in the very general sense of applying different marginal valuations to increments of consumption at different dates—is unavoidable in the utilitarian framework, and indeed in most other frameworks. One can however argue about the values of parameters that influence, but do not fully determine, the consumption discount rate and in particular determine whether that rate should be positive or negative—that is, whether future costs and benefits should be weighted less or more heavily than current costs and benefits when those costs and benefits are aggregated over time.

## SUMMARY: CONCLUSIONS AND RECOMMENDATIONS

This chapter provides an overview of economic valuation and the role it plays in the policy and environmental decision-making process. Although economic valuation does not capture all sources or types of value (e.g., intrinsic values on which the notion of rights is founded), it is much broader than usually presumed. It recognizes that economic value can stem from use of an environmental resource (use values), including both commercial and noncommercial uses, or from its existence even in the absence of use (nonuse value). The broad array of values included under this approach is captured by using the total economic value framework to identify potential sources of economic value. Use of this framework helps to provide a checklist of potential impacts and effects that must be considered in valuing ecosystem services as comprehensively as possible. It reduces the likelihood of omitting key sources of value, as well as the possibility of double counting values. By its nature, economic valuation involves the quantification of values based on a common metric, normally a monetary metric. The use of a dollar metric for quantifying values is based on the assumption that individuals are willing to trade the ecological service being valued for more of other goods and services represented by the metric (more dollars). Use of a monetary metric allows measurement of the costs or benefits associated with changes in ecosystem services.

The role of economic valuation in environmental decision-making depends on the specific criteria used to choose among policy alternatives. If policy choices are based primarily on intrinsic values, there is little need for the quantification of values through economic valuation. In such cases, the "benefit" of preservation is the protection of the right. In such cases, it may still be important to society to know how much protecting that right (e.g., preserving an intrinsically valuable endangered species) would cost—that is, what is being given up to ensure that protection, but there is no need to quantify the benefit of protection. However, if policymakers consider trade-offs and benefits and costs when making policy decisions, quantification of the value of ecosystem services is essential. Failure to include some measure of the value of ecosystem services in benefit-cost calculations will implicitly assign them a value of zero. The committee believes that considering the best available and most reliable information about the benefits of improvements in ecosystem services or the costs of ecosystem degradation will lead to improved environmental decision-making. The committee recognizes, however, that this information is likely to be only one of many possible considerations that influence policy choice.

The benefit and cost estimates that emerge from an economic valuation exercise will be influenced by the way in which the valuation question is framed. In particular, the estimates will depend on the delineation of the changes in ecosystem goods or services to be valued, the scope of the analysis (in terms of both the geographical boundaries and the inclusion of relevant stakeholders), and the temporal scale. In addition, the valuation question can be framed in terms of two alternative measures of value, willingness to pay and willingness to accept (compensation). These two approaches imply different presumptions about the distribution of property rights and can differ substantially, depending on the availability of substitutes and income limitations. In many contexts, methodological limitations necessitate the use of willingness to pay rather than willingness to accept.

Finally, because ecosystem changes are likely to have long-term impacts, some accounting of the timing of impacts is necessary. This can be done through discounting future costs and benefits. It is essential, however, to recognize that consumption discounting is distinct from the discounting of utility, which reflects the weights put on the well-being of different generations. When the impacts being valued are relatively limited, the discount rate that is used should be the consumption rate rather than the utility rate. The consumption discount rate can be positive or negative, depending on whether consumption is rising or falling. For environmental or ecological services that become scarcer over time, consumption would be falling, implying a negative discount rate.

Based on these conclusions, the committee provides the following recommendations:

- Policymakers should use economic valuation as a means of evaluating the trade-offs involved in environmental policy choices; that is, an assessment of benefits and costs should be part of the information set available to policymakers in choosing among alternatives.
- If the benefits and costs of a policy are evaluated, the benefits and costs associated with changes in ecosystem services should be included along with other impacts to ensure that ecosystem effects are adequately considered in policy evaluation.
- Economic valuation of changes in ecosystem services should be based on the comprehensive definition embodied in the total economic value framework; both use and nonuse values should be included.
- The valuation exercise should be framed properly. In particular, it should value the *changes* in ecosystem good or services attributable to a policy change. In addition, the scope should consider all relevant impacts and stakeholders, and the temporal scale of the analysis should be consistent with that of the impacts.
- The valuation exercise should indicate clearly whether (1) WTP or WTA measure of value was used, (2) in that context WTP is likely to differ significantly from WTA, (3) in that context WTP is likely to be strongly influenced by income differentials, and (4) use of the alternative value measure instead would likely have led to different policy prescriptions.
- In the aggregation of benefits and/or costs over time, the consumption discount rate, reflecting changes in scarcity over time, should be used instead of the utility discount rate.

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# 3
# Aquatic and Related Terrestrial Ecosystems

## INTRODUCTION

An ecosystem is generally accepted to be an interacting system of biota and its associated physical environment. Ecologists tend to think of these systems as identifiable at many different scales with boundaries selected to highlight internal and external interactions. In this sense, an aquatic ecosystem might be identified by the dominance of water in the internal structure and functions of an area. Such systems intuitively include streams, rivers, ponds, lakes, estuaries, and oceans. Most ecologists and environmental regulators also include vegetated wetlands as members of the set of aquatic ecosystems, and many think of groundwater aquifer systems as potential members of the set. "Aquatic and related terrestrial ecosystems" is a phrase that recognizes the impossibility of analyzing aquatic systems absent consideration of the linkages to adjacent terrestrial environments.

The inclusion of "related terrestrial ecosystems" for this study is a reflection of the state of the science that recognizes the multitude of processes linking terrestrial and aquatic systems. River ecologists have long understood the important connections between rivers and their floodplains (Junk et al., 1989; Stanford et al., 1996). The inflows of water, nutrients, and sediments from surrounding watersheds are heavily influenced by conditions within the floodplain. Conversely, floodplain plant and animal habitat value and sediment supply and fertility are often determined by river hydrology. This same sort of relationship between terrestrial and aquatic system is now understood to influence many of the functions of wetlands that motivate management efforts (Wetzel, 2001). Wetland ecologists have debated for years about appropriate recognition of capacity and opportunity to perform functions when conducting assessments of wetlands. A classic example of the discussion focuses on two identical wetlands, one in a pristine forested landscape, and the other in an intensely developed landscape. Both are assumed to have equivalent internal capacities to sequester pollutants, modify nutrient loads, and provide habitat, but the surrounding conditions mean that the opportunity for these functions to occur will differ significantly.

For many of the ecosystem functions and derived services considered in this chapter, it is not possible, necessary, or appropriate to delineate clear spatial boundaries between aquatic and related terrestrial systems (see Box 3-1). Indeed, to the extent that there is an identifiable boundary, it is often dynamic in both space and time. Floods, droughts, and seasonal patterns in rainfall are integral forcing functions for freshwater systems, just as tides, hurricanes, and sea-level rise constantly revise the boundaries between land and water in coastal systems. For these reasons, and as stated in Chapter 1, "aquatic ecosystems" collectively refers to aquatic and related terrestrial ecosystems unless noted otherwise.

The conceptual challenges of valuing ecosystem services involve explicit description and adequate assessment of the link (i.e., the ecological production function) between the structure and function of natural systems and the goods or services derived by humanity (see Figure 1-3). Describing structure is a relatively straightforward process, even in highly diverse ecosystems.

Exceptions sometimes arise at the levels of small invertebrates and microorganisms. However, function is often difficult to infer from observed structure in natural systems. Furthermore, the relationship between ecosystem structure and function as well as how these attributes respond to disturbance are not often well understood. Indeed, ecological investigations of aquatic systems show no signs of running out of questions about how these systems operate. Without comprehensive understanding of the behavior of aquatic systems, it is clearly difficult to describe thoroughly all of the services these systems provide society. Although valuing ecosystem services that are not completely understood is possible (see Chapters 4 and 5 for further information and examples), when valuation becomes an important input in environmental decision-making, there is the risk that the valuation may be incomplete.

There have only been a few attempts to develop explicit maps of the linkage between aquatic ecosystem structure/function and value. There are, however, a multitude of efforts to separately identify ecosystem functions, goods, services, values, and/or other elements in the linkage without developing a comprehensive argument. One consequence of this disconnect is a diverse literature that suffers somewhat from indistinct terminology, highly variable perspectives, and considerable divergent convictions. Despite these shortcomings, the core issue of how to assess and value aquatic ecosystem services is intuitive and important enough to support some synthesis—especially as related to environmental decision-making.

**BOX 3-1**
**Understanding Ecosystem Terminology**

Ecology is a scientific field that studies the relationships between and among (micro)organisms such as plants, animals, and bacteria and their environment. Like most scientists, ecologists use a variety of terms to describe aspects of their discipline. A few of the terms used throughout this report are defined below in the interest of facilitating the readability and understanding of this report.

***Ecosystem biodiversity*** describes a number and kinds of organisms in a specific geographic area that can be distinguished from other areas by its physical boundaries (e.g., lake, forest), though such boundaries can be somewhat arbitrary. In addition to biodiversity, ecosystems have properties such as the amount of plant and animal matter they produce (***primary and secondary production***) and the flow of chemical elements within and through the system (***nutrient cycling***).

***Ecosystem structure*** refers to both the composition of the ecosystem (i.e., its various parts) and the physical and the biological organization defining how those parts are organized. A leopard frog or a marsh plant such as a cattail, for example, would be considered a component of an aquatic ecosystem and hence part of its structure. The relationship between primary and secondary production would also be part of the ecosystem structure, because it reflects the organization of the parts.

***Ecosystem function*** describes a process that takes place in an ecosystem as a result of the interactions of plants, animals, and other (micro)organisms in the ecosystem with each other or their environment and that serves some purpose. Primary production (most notably the generation of plant material) is an example of an ecosystem function. The ***net primary production*** in an ecosystem is determined by the number and kinds of plants present; the amounts of sunlight, nutrients, and water available; and the amount of this productivity used internally by the plants themselves.

Ecosystem structure and function provide various ***ecosystem goods and services*** to humans that have value: for example, rare species of plants or animals, fish for recreational or commercial use, clean water to swim in or drink. The functioning of ecosystems (interaction of organisms and the physical environment) often provides for services such as water purification, recharge of groundwater, flood control, and various aesthetic qualities such as pristine mountain streams or wilderness areas.

The goal of this chapter is to review and summarize some of the common elements in the published literature concerning the identification of aquatic ecosystem functions and their linkage to goods and services for subsequent economic valuation. It also includes a summary review of the extent and status of aquatic ecosystems in the United States and some of the issues that continue to complicate efforts to value aquatic ecosystem services. The chapter closes with a summary of its conclusions and recommendations.

## EXTENT AND STATUS OF AQUATIC AND RELATED TERRESTRIAL ECOSYSTEMS IN THE UNITED STATES

There are impressive examples of almost every kind of aquatic ecosystem within the United States. The country has some of the largest freshwater lakes in the world (see Box 3-2), one of the world's largest river systems (see Box 3-3), one of the world's largest estuaries (see Box 3-4), thousands of miles of coastline, extensive underground aquifers (see Box 3-5), a vast array of tidal and nontidal wetlands (see Box 3-6), and so many small creeks and streams that they are still being mapped. There is a long history of efforts to understand and manage these resources for public and private benefit, and the need to make informed decisions continues to motivate both research and monitoring. These short summaries identify some of the ways that humans have used and benefited from these ecosystems over time and many of the ecosystem services that managers seek to value in efforts to inform decisions. The summaries also identify some of the key management issues that have arisen as a result of evolving and often conflicting interests regarding ecosystem services.

In 2002, U.S. Environmental Protection Agency (EPA) released the *2000 National Water Quality Inventory* (NWQI; EPA, 2002)—the thirteenth installment in a series that began in 1975. These reports are required by Section 305(b) of the Clean Water Act and are considered by EPA to be the primary vehicle for informing Congress and the public about general water quality conditions in the United States. As such, the reports characterize water quality, identify widespread water quality problems of national significance, and describe various programs implemented to restore and protect U.S. waters. Notably, these assessments include streams and rivers, lakes and ponds, coastal resources to include tidal estuaries, shoreline waters (coastal and Great Lakes), and wetlands. Table 3-1 summarizes some of the relevant results and findings from the 2002 NWQI report.[1]

Although EPA, various federal and state partners, and other nongovernmental organizations and scientists have been assessing the condition of estuaries for decades, the *National Coastal Condition Report* (NCCR; EPA, 2001) represents the first comprehensive summary of coastal conditions in the United States and uses data and information collected from 1990 to 2000.[2] The report, a coordinated effort between EPA (lead) the National Oceanic and Atmospheric Administration (NOAA), the U.S. Geological Survey (USGS), and the U.S. Fish and Wildlife Service (USFWS), compiles and summarizes several data sets from federal and

[1] The NWQI report includes information about water quality standards, detailed summaries of the results of waterbody assessments by designated uses and states, and a discussion of the data collection and analysis methods used in that report.

[2] Interested readers are directed to the NCCR report (EPA, 2001) for further information and details on the findings as well as data collection and analysis methods used to generate and interpret the regional results. Notably, Chapter 1 of that report also includes a comprehensive list of federal programs and initiatives that address coastal issues, many of which are conducted jointly with various coastal states and local organizations.

**BOX 3-2**

**Great Lakes Ecosystem**

The Great Lakes ecosystem is the largest freshwater system in the world, comprising Lakes Michigan, Superior, Huron, Erie, and Ontario. Collectively, they cover a land area of 94,000 square miles and contain 5,500 cubic miles of water in the United States and Canada. Rivers and streams running into the lakes drain 201,000 square miles of land. Rain that falls in Chicago or Duluth may eventually leave the ecosystem more than 1,000 water miles to the east at Montreal, although outflows of water and its solutes are small, less than 1 percent by volume per year.

Habitats within the ecosystem are diverse. In the north, forests surrounding Lake Superior support healthy populations of black bears, bald eagles, wolves, and moose. Waterfowl, songbirds, and raptors funnel between Lakes Michigan and Erie during the spring and fall migrations. Lakes, wetlands, and uplands across the basin provide a mixture of habitats for temperate plants and animals of many types. The beaches and dunes of the southern shores are nesting areas for open water birds and wading birds such as the endangered piping plover.

Mining, timbering, agriculture, and industry brought major changes to the ecosystem beginning in the 1800s. Industries of all sorts grew up on the shorelines of lakes and rivers and used these waterbodies to facilitate both waste disposal and shipping. New locks and canals between the lakes allowed access to the Atlantic, while also opening pathways for the introduction of exotic species. For example, saltwater alewives displaced native species and sea lamprey devastated Great Lakes trout populations. Although industry created great wealth and well-being, it also left behind vast quantities of waste, including residues of dichlorodiphenyltrichloroethane and 1,1,1-trichloro-2,2-bis(4-chlorophenyl)ethane (DDT), polychlorinated biphenyls (PCBs), and heavy metals. Sewage and soil erosion turned lake water from clear blue to thick green through eutrophication.

Different trends began in the 1960s. Economic and public policy changes began to stem the flow of pollutants into the system, while aging mines, mills, and refineries closed. Electricity and natural gas replaced coal for heating, and air pollution laws cut power plant and automobile emissions. DDT and PCBs were banned, and the use of heavy metals declined. Treaties with Canada and interstate agreements established ecosystem-wide authorities to identify environmental problems and implement solutions. Marked changes in the former ecosystem followed these economic and regulatory changes. Water quality gradually improved so that the "oligotrophic blue" is reestablished in all the lakes. Between 1974 and 1994, PCB levels in top-of-the-food-web predators dropped by as much as 90 percent. Bald eagles once again breed along lake and river shorelines, and shoreline beaches and dunes are major summer destinations. Boating and recreational fishing are multibillion dollar industries.

However, history and the daily activities of 33 million people present continuing challenges for the ecosystem. Old harbors and shipping points are still lined by millions of tons of toxic materials and sediments. Although ambient concentrations are low, persistent toxic materials are concentrated by the ecosystem and food web, and levels of metals and PCBs in the blood and tissue of fish, waterfowl, and birds of prey are still high. Fish consumption advisories for recreational anglers remain in effect in across the region, and further reductions in mercury use and emissions remain a regulatory priority.

Restoring habitat and native species is also a priority. Wetland regulations halted the destruction of rare wetland types such as cedar bogs, fens, and salt marshes. Wetland restoration aims at restoring scarce wetland types, especially those along Great Lakes shorelines and bird migration routes. Elk and moose are reestablished in some areas, and significant efforts are under way to strengthen populations of Lake Superior native clams, walleye, brook trout, and sturgeons. Invasive and exotic species such as zebra mussels, lamprey, ruffe, and goby, however, continue to displace and threaten native species.

The Great Lakes region can be viewed a continuing experiment in testing human capability to live and prosper within the bounds of a major aquatic ecosystem, and although the last four decades allow some optimism, major environmental problems remain. During storms, combined sewer and stormwater drainage systems overflow, releasing untreated sewage in otherwise protected waterbodies. Urban and agricultural runoff contribute excessive nutrients into susceptible bays and inlets. Toxic air emissions disperse trace contaminants across the region, feeding the cycle of bioaccumulation. Success in this Great Lakes experiment will not be accidental. Thus, careful choices must be made and subsequent actions taken.

SOURCE: Great Lakes National Program Office (2001, 2002).

**BOX 3-3**
**The Missouri River Ecosystem**

The Missouri River basin extends over 530,000 square miles and covers approximately one-sixth of the continental United States. The one-hundredth meridian, the widely accepted boundary between the arid western states and the more humid states in the eastern United States, crosses the middle of the basin. The Missouri River's source streams are in the Bitterroot Mountains of northwestern Wyoming and southwestern Montana. The Missouri River begins at Three Forks, Montana, where the Gallatin, Jefferson, and Madison Rivers merge on a low, alluvial plain. From there, the river flows to the east and southeast to its confluence with the Mississippi River just above St. Louis. Near the end of the nineteenth century, the Missouri River's length was measured at 2,546 miles.

Between 1804 and 1806, the famous explorers Meriwether Lewis and William Clark led the first recorded upstream expedition from the river's mouth at St. Louis to the Three Forks of the Missouri, and eventually reached the Pacific coast via the Columbia River. The Missouri River subsequently became a corridor for exploration, settlement, and commerce in the nineteenth and early twentieth centuries, as navigation extended upstream from St. Louis to Fort Benton, Montana. Social values and goals in the Missouri River basin during this period reflected national trends and the preferences of basin inhabitants. Statehood, federalism, and regional demands to develop and control the river produced a physical and institutional setting that generated demands from a wide range of interests.

The Missouri River ecosystem experienced a marked ecological transformation during the twentieth century. At the beginning of the century, the Missouri River was notorious for large floods, a sinuous and meandering river channel that moved freely across its floodplain, and massive sediment transport. However, by the end of the twentieth century, the Missouri River bore little resemblance to the previously wild, free-flowing river. Over time, demands for the benefits associated with the Missouri's control and management resulted in significant and lasting physical and hydrologic modifications of the river. These modifications led to substantial changes in the river and floodplain ecosystem. Numerous reservoirs are scattered across the basin, with seven large dams and reservoirs located on the river's mainstem.

Ecological changes that accompanied changes in hydrology proceeded more slowly but were of a similar magnitude. Large floodplain areas along the upper Missouri were inundated by the reservoirs, and large areas of native vegetation communities in downstream floodplains were converted into farmland. Many native fish and avian species experienced substantial reductions, while nonnative species—especially fish—thrived in some areas. The rich biodiversity of the pre-regulated Missouri River ecosystem was sustained through a regime of natural disturbances that included periodic floods and attendant sediment erosion and deposition. These disturbances, in turn, supported a variety of ecological benefits, including commercial and recreational fishing, timber, wild game, trapping and fur production, clean water, soil replenishment processes, and natural recharge of groundwater. Flow regulation and channelization substantially changed the Missouri River's historic hydrologic and geomorphic regimes. The isolation of the Missouri River from its floodplain caused by river regulation structures has in many stretches largely eliminated the flood pulse and its ecological functions and services. As a result of these changes, the production and the diversity of the ecosystem have both markedly declined.

For purposes of comparison, the major benefits of river regulation come from hydropower, water supply, and flood damage reduction, each of which has annual benefits measured in hundreds of millions of dollars. Recreation comes next, with annual benefits measured in tens of millions of dollars. Navigation follows, with annual benefits measured in millions of dollars. The value of ecosystem services that have been forgone in order to achieve other benefits is largely unknown.

Today the Missouri River floodplain ecosystem consists of extensive ecosystems in and around the large reservoirs, open reaches of channel, and riparian floodplains. Some of these systems are recognized producers of recreational opportunities or agriculture. Some traditional ecosystems, particularly those representing the historical habitats of the pre-regulated Missouri, have been less well recognized for the social values provided through ecosystem services. Many ecosystem services, such as fish, game, and aesthetic values, are not monetized and are not traded in markets. They thus tend to be underappreciated and undervalued by the public and by decision-makers.

SOURCE: NRC (2002b).

**BOX 3-4**
**Chesapeake Bay**

The Chesapeake Bay is the largest estuary in the United States and among the largest in the world. The watershed spreads over approximately 64,000 square miles, encompassing major portions of Pennsylvania, Maryland, and Virginia; all of the District of Columbia; and lesser portions of New York, West Virginia, and Delaware. It receives freshwater from six major rivers and has more than 2,000 square miles of relatively protected tidal waters.

The bay has been prized by its human inhabitants for centuries for its ability to provide food, water, navigation, waste disposal, recreation, and aesthetic pleasures. The estuary supports extensive commercial and recreational fisheries for striped bass, menhaden, flounder, perch, and many others. Oyster, crab, and clam harvests have supported local fishermen for generations. In addition, important habitat is provided for sea turtles, sharks, rays, eels, whelks, and an enormous diversity of waterfowl.

Hampton Roads located at the mouth of the bay in Virginia and Baltimore near the head of the bay in Maryland are among the nation's largest ports. Hampton Roads is home to the world's largest naval base, and both ports contain major international shipping terminals. Shipbuilding and repair are major industries in the regional economy. The value of commercial navigation in the bay is rivaled by the tremendous investment in recreational boating that operates from hundreds of marinas and thousands of private docks. The more than 20,000 miles of tidal shoreline in the system also provide highly desired home locations for many of the area's residents.

All of these benefits have led to intensive and continually increasing pressure on the ecosystem as human populations in the region have increased and subsequent use has escalated. One consequence has been emergence of the Chesapeake Bay as one of the most extensively studied estuaries in the world. Interest in the system has been driven by concern for declines in finfish and shellfish populations. These trends are recognized as the result of overharvesting, pollution, habitat destruction, and introduced diseases. The challenge of restoring the system's productivity has motivated investment of millions of dollars of public funds through the Chesapeake Bay Program, a cooperative effort by states and the federal government to reduce impacts and improve conditions in the ecosystem. The extensive and complex array of stakeholder groups, commitments, and programs orchestrated under the umbrella of this program has become a model for similar efforts emerging in other large aquatic ecosystems.

The current focus of the Chesapeake Bay Program is on reduction of nutrient, sediment, and toxic inputs to the system. This is being accomplished through the use of state-of-the-art simulation models, extensive monitoring, outreach and education, and a mix of regulatory and nonregulatory programs to design and implement best management practices throughout the watershed. Parallel efforts are under way to restore vital habitats such as wetlands, submerged aquatic vegetation, and oyster reefs; promulgate multispecies and ecosystem management plans; and control the impacts of continuing development.

Estimates of the funding necessary to achieve restoration goals in the Chesapeake Bay extend into the tens of billions of dollars. This amount exceeds currently available resources by several orders of magnitude, creating unavoidable need to prioritize such efforts. To date, the incorporation of economic valuation in bay program management has been informal. Although cost-benefit analyses are implicit in almost every budget decision for Program activities, explicit use of economic assessments is not a characteristic of program management.

SOURCE: Scientific and Technical Advisory Committee (2003).

**BOX 3-5**
**The Edwards Aquifer and Groundwater Recharge in San Antonio, Texas**

The Edwards Aquifer of central Texas is a highly permeable karst limestone on the edge of the Chihuahuan Desert. The average annual temperature is 20.5°C average annual precipitation is 28.82 inches. The annual recharge for the aquifer ranges from 44,000 to 2,000,000 acre-feet and averages 635,500 acre-feet per year. Thousands of springs flow from this groundwater source, including the largest springs in the state, and potable water is the primary use of the groundwater supply (Bowles and Arsuffi, 1993). Recharge of the aquifer has been monitored by the U.S. Geological Survey (USGS) since 1915, while water quality monitoring began in 1930.

Currently, more than 1.7 million people rely on the Edwards Aquifer. However, recharge of the porous karstic limestone occurs primarily during wet years when precipitation infiltrates deeply into the soils and underlying rock. As a result, new laws were introduced that changed the legal basis of ownership from "right of capture" for a demonstrated "beneficial use" of the extracted water to a new approach based on prior appropriation (i.e., senior water rights). Concern increased as several springs (Comal, San Antonio, San Pedro) in the area began to dry up following a seven-year drought in the 1950s. Groundwater storage is critical in most aquatic ecosystems to provide persistent springs and streams during drought. Diverse microbial communities and a wide range of invertebrate and vertebrate species live in groundwaters (Gibert et al., 1994; Jones and Mulholland, 2000). Their main ecosystem functions are breaking down organic matter and turning dead materials (detritus) into live biomass that is consumed in food webs. Thus, these species recycle nutrients and are important in secondary productivity. The trade-offs in extracting groundwater include possible loss of habitat for endemic species that are protected by state and federal regulations. For example, the Edwards Aquifer-Comal Springs ecosystem provides critical habitat for the Texas blind salamander (Crowe and Sharp, 1997; Edwards et al., 1989). Moreover, 91 species and subspecies of fish are endemic in this underground ecosystem (Bowles and Arsuffi, 1993; Culver et al., 2000; Longley, 1986). Several economic values of groundwater are associated with ecosystem services such as processing of organic matter by diverse microbes and invertebrates, providing possible dilution of some types of surface-originating contaminants, and sustaining populations of rare and endangered species that are often restricted to very local habitats (Culver et al., 2000).

By 1970, new regulations were issued to protect water quality in the Edwards Aquifer. These new rules limited economic development within the recharge zone to balance the long-term average recharge rate with the extraction rate. This steady-state equilibrium, however, is often characterized by time lags in recharge and drought frequencies that complicate predictable levels of water supply. Other physical considerations include how much and what types of development occur without disrupting rapid infiltration of the recharge zone. Degradation of subsurface water quality as well as declines in rates of recharge occur when economic development increases the extent of impervious surfaces that, in turn, cause more rapid runoff and loss of infiltration during and after precipitation events. The increased surface area of roof tops, roads, parking lots, and so on changes stormwater and groundwater hydrology and water chemistry. As groundwater is depleted the cost for deeper drilling and pumping increases costs and can terminate or slow the rate of extraction. Thus, it is difficult to consistently define "overextraction." The rate of extraction depends on future values relative to current values under specific alternative uses and climatic conditions (Custodio, 2002).

The Texas legislature created the Edwards Aquifer Authority to control pumping and to reallocate water through market mechanisms (Kaiser and Phillips, 1998; McCarl et al., 1999; Schaible et al., 1999). This approach has reallocated water from lower economic uses (e.g., agricultural irrigation) to higher-valued uses (e.g., for domestic and industrial water supplies and environmental and recreational uses). Especially during dry years, it appears feasible for transfers from irrigation to offset demands for municipal water supplies. In 1997, farmers accepted an offer of $90 per acre prior to the cropping season in a pilot study of the Irrigation Suspension Program (Keplinger and McCarl, 2000; Keplinger et al., 1998). Drought increases the demand for water while the supply declines. Chen et al. (2001) used a climate change model to estimate the regional loss of welfare at $2.2 million to $6.8 million per year from prolonged drought. To protect endangered species in springs and groundwater, an additional reduction of 9 to 20 percent in pumping would add $0.5 million to $2 million in costs.

*continues*

**BOX 3-5 Continued**

Traditionally, the only costs for the use of groundwater was the expense of installing a well and paying for pumping of this "open-access, free resource." However, when rates of extraction exceed recharge, the reduction in water levels may exceed an uncertain threshold, and cause irreversible changes. For example, removal of water in the underground area may cause collapse of the overlying substrata. These collapses decrease future storage capacity below ground and can alter land values. In some areas the depleted groundwater may cause intrusion of low-quality water from other aquifers or from marine-derived salt or brackish waters that could not readily be restored for freshwater storage and use. Contamination of groundwater from landfills, leaking petroleum storage tanks, and pesticides can also makes aquifers unusable.

In 1993 the Sierra Club sued the state for failure to guarantee a minimum flow of 100 cubic feet per second (cfs) to Comal and San Marcos Springs. The State of Texas and the U.S. Fish and Wildlife Service have entered into an agreement to resolve this conflict. To avoid jeopardizing the endangered species living in these springs, the Edwards Aquifer Authority banned the use of irrigation sprinklers whenever flow declined below a threshold that limited habitat in the Comal Springs. Approximately 1.5 million people were affected when the USGS reported that the flow declined to 145 cfs in September 2002. Limited pumping also had large economic consequences on agriculture. While water markets may ultimately resolve reallocation issues among stakeholders in the Edwards Aquifer region (Chang and Griffin, 1992; Kaiser and Phillips, 1998; McCarl et al., 1999; Schaible et al., 1999), the predictability of water markets as suppliers of water for different needs is complex and will help reallocate water only if some level of supply is available.

The construction of water-transfer pipelines and additional surface storage reservoirs is under consideration along with conjunctive storage (pumping water into sub-surface storage associated with aquifers.) The estimated cost of building a surface reservoir (Applewhite) to provide an additional 170,000 acre-feet of water for sale was $317 per acre-foot compared to $67 per acre-foot if pumped from the Edwards Aquifer (John Merrifield, University of Texas-San Antonio, personal communication, 2003). The combination of climatic change (more extremes in drought and in distribution of rainfall) and increased human population growth will stress the current rules on allocation of water to maintain natural ecosystem functions and survival of endangered species.

**BOX 3-6**
**The South Florida Ecosystem**

South Florida is dominated by the waters of the Kissimmee-Ockeechobee-Everglades (KOE) ecosystem. In the late summer and fall, rainfall enters the Kissimmee River near Orlando and gradually flows south to Lake Ockeechobee. The waters gather more rainfall and continue south, flowing into agricultural fields, an extensive system of flood control canals and reservoirs, and the river of grass called the Everglades. Eventually, the waters flow through the Everglades to enrich the mangrove forests and estuaries on the Atlantic and Gulf Coasts (Purdum, 2002).

The KOE ecosystem covers almost 17,000 square miles in South Florida. The ecosystem is home to more than 6 million people and the dynamic regional economies of Orlando and South Florida, including the cities of Miami, Fort Lauderdale, and West Palm Beach. The ecosystem's preserves and natural areas are known throughout the world for their uniqueness and beauty: including the Everglades National Park, Big Cypress Preserve, the Florida Keys, Biscayne Bay, and the estuary of Florida Bay (NRC, 2002a, 2003).

The ecosystem is a mix of natural and human forces. Ten thousand years ago, the KOE area was dry prairie, inhabited by horses, camels, bison, and mammoths and the humans who hunted them. About 9,000 years ago, the oceans began to rise with the ending of the last ice age. The habitat shifted as the climate changed to humid subtropics in the north and tropical savannah in the south (Purdum, 2002). Swamps, marshes, pinelands, the everglades, and hardwood hammocks developed in inland areas, sustained by the gradual flow of waters. Mangroves and estuaries gained a footing in coastal areas. Tropical and subtropical wildlife grew in abundance, ranging from crocodiles to bear to birds in wide variety.

*continues*

**BOX 3-6 Continued**

In the last 100 years, the annual tropical cycle of sun in the winter drought and dependable rain in the summer and fall attracted residents from around the world, but torrential rains caused flooding. As settlements grew, there was a steady human effort to control and redirect the annual flooding. Some redirected water went to serve urban and agricultural uses, but much was simply channeled into the ocean.

By the end of the twentieth century, the KOE ecosystem was crisscrossed by more than 1,800 miles of canals and levees, controlling the floods but also cutting off the established flows of KOE water. Water became scarce in humid area such as the Everglades and Florida Bay estuaries. Some species were particularly hard hit. Nesting wading birds declined by 90 percent (Lord, 1993). Saltwater began to intrude into freshwater aquifers supplying 90 percent of potable water for the human population (Purdum, 2002).

Major investments are now being made to restore the quantity of water available and its flow through the remaining natural systems. One significant project is the $7.8 billion Everglades Restoration Plan (see NRC, 2002a; 2003). The plan proposes to remove major barriers to water flows into Everglades National Park, treat surface water runoff from urban areas, reuse wastewater, and store water from heavy rainfall rather than shunting it out to sea (Purdum, 2002). The project is expensive, but is it enough given the value of ecosystem resources and services? Methods for valuing ecosystem services would help provide an answer.

state coastal monitoring programs to present a broad baseline picture of the condition of U.S. coastal waters as divided into five discrete regions: Northeast, Southeast, Great Lakes, Gulf of Mexico, and West Coast. The report is intended to serve as a benchmark for assessing the progress of coastal programs in the future and will be followed by subsequent reports on more specialized coastal issues. It is important to note that the condition of U.S. coastal waters is described primarily in terms of data on estuaries, which are loosely defined in the NCCR as the productive transition areas between freshwater rivers and the ocean. In addition, although the intent of the report is to evaluate the condition of coastal waters (i.e., primarily estuaries) nationwide, the report states that there was insufficient information to completely assess West Coast estuaries and the Great Lakes, and no assessment was possible for the estuarine systems of Alaska, Hawaii, and other island territories. However, new ecological programs, both newly created and proposed, should permit a comprehensive and consistent assessment of all of the nation's coastal resources by 2005. The NCCR used aggregate scores for a total of seven water quality indicators (water clarity, dissolved oxygen, coastal wetland loss, contaminated sediments, benthos, fish tissue contaminants, and eutrophic condition); 56 percent of assessed estuarine areas (representing more than 70 percent of the estuarine areas of the conterminous United States, excluding Alaska) were found to be in good condition for supporting aquatic life use (plant and animal communities) and human uses (e.g., water supply, recreation, agriculture). In contrast, 44 percent of the nation's estuaries were characterized as impaired for human use (10 percent), aquatic life use (11 percent), or both (23 percent). In general, the nation's coastal areas were rated as poor if the mean conditions for the seven indicators showed that more than 20 percent of the estuarine area in that region was degraded.

Section 401 of the Emergency Wetlands Resources Act of 1986 requires the USFWS to conduct studies of the status and trends of the nation's wetlands and report the results to Congress each decade. The third report of the USFWS's National Wetlands Inventory (NWI), *Status and Trends of the Wetlands in the Conterminous United States 1986 to 1997*, was released

TABLE 3-1 Selected Findings and Results from the 2002 National Water Quality Resource Inventory

| Waterbody Type | Total Size[a] | Amount[b] Assessed (% of total) | Good[c] (% of assessed) | Impaired[d] (% of assessed) | Leading Pollutants and Causes of Impairment[e] | Leading Sources of Impairment[e] |
|---|---|---|---|---|---|---|
| Rivers and streams | 3,692,830 miles | 699,946 miles (19%) | 426,633 miles (61%) | 269,258 miles (39%) | Pathogens (bacteria)<br>Siltation<br>Habitat alteration<br>Oxygen-depleting substances<br>Nutrients<br>Thermal modification<br>Metals<br>Flow alteration | Agriculture<br>Hydrologic modification<br>Urban runoff and storm sewers<br>Forestry<br>Municipal point sources<br>Resource extraction |
| Lakes, reservoirs, and ponds | 40,603,893 acres | 17,339,080 acres (43%) | 9,375,891 acres (55%) | 7,702,370 acres (45%) | Nutrients<br>Metals<br>Siltation<br>Total dissolved solids<br>Oxygen-depleting substances<br>Excess algal growth<br>Pesticides | Agriculture<br>Hydrologic modification<br>Urban runoff and storm sewers<br>Atmospheric deposition<br>Municipal point sources<br>Land disposal |
| Coastal resources: Estuaries | 87,369 sq. miles | 31,072 sq. miles (36%) | 14,873 sq. miles (49%) | 15,676 sq. miles (51%) | Metals<br>Pesticides<br>Oxygen-depleting substances<br>Pathogens (bacteria)<br>Priority toxic organic chemicals<br>Polychlorinated biphenyls (PCBs)<br>Total dissolved solids | Municipal point sources<br>Urban runoff/storm sew<br>Industrial discharges<br>Atmospheric deposition<br>Agriculture<br>Hydrologic modification<br>Resource extraction |

| Waterbody Type | Total Size[a] | Amount[b] Assessed (% of total) | Good[c] (% of assessed) | Impaired[d] (% of assessed) | Leading Pollutants and Causes of Impairment[e] | Leading Sources of Impairment[e] |
|---|---|---|---|---|---|---|
| Coastal resources: Great Lakes shoreline | 5,521 miles | 5,066 miles (92%) | 1,095 miles (22%) | 3,955 miles (78%) | Priority toxic organic chemicals<br>Nutrients<br>Pathogens (bacteria)<br>Sedimentation and Siltation<br>Oxygen-depleting substances<br>Taste and odor<br>PCBs | Contaminated sedimen<br>Urban runoff and storm sewers<br>Agriculture<br>Atmospheric deposition<br>Habitat modification<br>Land disposal<br>Septic tanks |
| Coastal resources: Ocean shoreline waters | 58,618 miles | 3,221 miles (6%) | 2,755 miles (86%) | 434 miles (14%) | Pathogens (bacteria)<br>Oxygen-depleting substances<br>Turbidity<br>Suspended solids<br>Oil and grease<br>Metals<br>Nutrients | Urban runoff and storm sewers<br>Nonpoint sources<br>Land disposal<br>Septic tanks<br>Municipal point sources<br>Industrial discharges<br>Construction |
| Wetlands | 105,500,000 acres[f] | 8,282,133 acres (8%) | 4,839,148 acres (58%) | 3,442,985 acres (42%) | Sedimentation and siltation<br>Flow alterations<br>Nutrients<br>Filling and draining<br>Habitat alterations<br>Metals | Agriculture<br>Construction<br>Hydrologic modification<br>Urban runoff<br>Silviculture<br>Habitat modifications |

[a] Units are miles for rivers and streams; acres for lakes, reservoirs, ponds, and wetlands; square (sq.) miles for coastal Lake shoreline, and ocean shoreline waters).
[b] Includes waterbodies assessed as not attainable for one or more designated uses (i.e., total number of waterbody uni impaired do not necessarily add up to total assessed).
[c] Fully supporting all designated uses or fully supporting all uses, but threatened for one or more uses.
[d] Partially or not supporting one or more designated uses.
[e] For those states and jurisdictions that reported this type of information (i.e., often a subset of the total number of state assessed and reported on various waterbodies; see EPA 2002 for further information).
[f] From *Status and Trends of Wetlands in the Conterminous United States 1986 to 1997* (Dahl, 2000).
SOURCE: Adapted from EPA (2002).

in 2000 (Dahl, 2000). This NWI report provides the most recent and comprehensive estimates of the areal extent (status) and trends of wetlands in the conterminous 48 United States on all public and private lands between 1986 and 1997. In that report, wetlands, deepwater, and upland (land-use) categories are divided into a wide variety of habitats and groupings; however, wetlands are classified principally as estuarine and marine wetlands and freshwater wetlands.[3] The study design included 4,375 randomly selected sample plots 4 square miles in area that were examined using remotely sensed data in conjunction with fieldwork and verification to determine wetland change. However, the report does not address water quality conditions or provide an assessment of wetland functions.

As of 1997, the lower 48 states contained about 105.5 million acres of wetlands of all types (Dahl, 2000),an area about the size of California. Of these, about 95 percent are inland freshwater wetlands, while the remaining 5 percent are saltwater (marine and estuarine) wetlands. Between 1986 and 1997, the net loss of wetlands was 644,000 acres with an annual loss rate of 58,545 acres (see also Table 1-1); 98 percent of these losses occurred in freshwater wetlands.[4]

A fourth major federal program report related to the extent and status of aquatic and related terrestrial ecosystems is the *Summary Report of the 1997 National Resources Inventory (revised December 2000)* (USDA, 2000). The NRI is conducted every five years by the U.S. Department of Agriculture's Natural Resources Conservation Service in cooperation with the Iowa State University Statistical Laboratory. The 1997 NRI report is the fourth summary report in a series that began in 1982 and is a scientifically based, longitudinal panel survey designed to consistently assess conditions and trends of the nation's soil, water, and related resources for all nonfederal lands for all 50 states and other jurisdictions (e.g., Puerto Rico) using photo interpretation and other remote sensing methods and techniques. Thus, all values provided in the 1997 NRI report are estimates based on data collected at sample sites, not data taken from a census.[5]

## CATALOGING ECOSYSTEM STRUCTURE AND FUNCTION AND MAPPING ECOSYSTEM GOODS AND SERVICES

### Ecosystem Structure and Function

As a general rule, the literature on ecosystem valuation attempts to use the terms "structure" and "function" as descriptors of natural systems (i.e., free of "value" content; see Chapter 2 for further discussion). These are features of natural systems that result in a capacity to provide goods and services, which can in turn be valued by humans (see also Box 3-7). The "value-free" distinction is ultimately blurred when considering intrinsic values of natural systems, but identification of ecosystem structure and function is a reasonable starting point for the subsequent mapping of ecosystem goods and services.

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[3]See Table 1 and Appendixes A through B in Dahl, 2000 for further information.
[4]This and other USFW's NWI reports, their data, resources, and other information are available on-line at
[5]*http://wetlands.fws.gov.* Accessed June 11, 2004.
The 1997 NRI report has detailed information on study design, data collection methods, compilation, synthesis, and analysis, in addition to the resource inventory results.

**BOX 3-7**
**Energy Analysis and Valuation**

Some ecologists use energetics (Odum, 1988, 1996) as a common currency for valuation. More specifically, energetic valuation (Odum and Odum, 2000) attempts to put the contributions of the economy on the same basis as the work of the environment by using *one kind* of energy (e.g., solar energy) as the common denominator. Accordingly, the term "emergy" was proposed to express all values in *one kind* of energy required to produce designated goods and services, for the purpose of eliminating confusion with other energetic valuation concepts (Odum, 1996). As an example, to evaluate the total worth of an estuary, the total energy flow in terms of embodied energy (which represents all of the work of the ecosystem) is determined and then this energy value is converted to monetary units on the basis of the ratio between energy and money in the production of market goods (Odum, 1993).

Energetic evaluation is presented as a strategy by which ecological data can be used to influence environmental policies (Odum and Odum, 2000) and it has served as a useful tool to examine the interface between ecosystems and economics (e.g., Odum and Turner 1990; Turner et al. 1988). However, it rejects the premise that values arise from the preferences of individuals and that the fundamental purpose of economic valuation is to estimate the change in willingness to pay (or accept) for the various losses and gains experienced by individuals when confronted by changes in ecosystem services.

There are at least three key elements in the effective description of aquatic ecosystems: (1) geomorphology, (2) hydrology, and (3) biology. Collectively, these factors constrain the stocks of organic and inorganic materials in the system and the internal and external fluxes of those materials and energy. For this reason, many classification efforts focus on these three elements in developing taxonomies of aquatic ecosystems.

An example of extant classification systems is the one adopted by the NWI of the USFWS (Cowardin et al., 1979). This hierarchical system distinguishes general kinds of aquatic ecosystems (e.g., rivers, lakes, estuaries) and then places special emphasis on a site's vegetative community and hydroperiod. The method does not purport to address function. Indeed, much of the relevant literature in wetlands ecology documents the great variability of functions within and among NWI wetland types.

A newer classification scheme developed by Brinson (1993), called the HydroGeomorphic Method (HGM) is now being developed into an assessment methodology by the U.S. Army Corps of Engineers and the EPA (Smith et al., 1995). The HGM classification places emphasis on the hydrology and topographic setting of a wetland. The classification system has become the basis for development of a growing number of wetland condition assessment models. The models support evaluation of the degree of departure from ideal or "reference" conditions for specific classes of wetlands. The assumption is that stressors in the wetland or surrounding landscape (e.g., soil disturbance, grazing, pollution discharges) will affect the natural functions of the ecosystem and that this effect can be related to observable changes in the wetland. This approach begins to establish a relationship between wetland condition and capacity to perform certain functions. Nevertheless, the natural variability of wetland ecosystems confounds simple inference about functions based simply on HGM classification.

There are similar efforts to develop classifications for lakes (e.g., Busch and Sly, 1992; Maxwell et al., 1995) and streams (e.g., Rosgen, 1994; TNC, 1997; Vannote et al., 1980). Again, each of these approaches starts with structural attributes of the system being evaluated and directly or indirectly addresses some aspect(s) of function. However, none of these efforts purport to support direct inferences about a comprehensive suite of ecological functions.

The fact that there is no explicit and invariant link between structure and function of aquatic ecosystems is part of the problem in efforts to assess all goods and services provided by these natural systems. If the behavior of a particular ecosystem is dependent not only on its composition, but also on linkages to surrounding systems and the impact of stressors, then comprehensive recognition of goods and services provided is not straightforward. The constantly evolving body of work on wetlands assessment exemplifies this challenge. Describing the structure of wetland ecosystems in terms of plant community composition, soil characteristics, and water movement is a well-developed practice with generally accepted protocols. Assessing the level of function in a wetland is, however, an exceptionally complex undertaking. As noted previously, a wetland's "capacity" to perform a function interacts with its "opportunity" to perform the function.

In a simple example case of habitat function, the structural characteristics of a wetland determine its capacity to meet the requirements of amphibians. The amounts of open water, the seasonal patterns of soil saturation, the types of sheltering plant material, and the size of the wetland all combine to determine if the wetland could support amphibians (e.g., Sousa, 1985). Landscape setting, or the larger system within which the wetland system exists, determines other factors that affect a wetland's opportunity to reach its potential as amphibian habitat. Adjacent land use affects access, water quality, and the density of potential predator populations. These and other external factors have significant impacts on the level at which habitat functions are performed (e.g., Knutson et al., 1999). The point is that wetland ecosystem structure alone is not an adequate predictor of the amphibian habitat services provided. Thus, as a generality, mapping ecosystem goods and services does not proceed linearly from system structure.

The default response to the lack of a simple logic linking structure to function has been development of generalized lists of potential functions appropriate to broad categories of aquatic ecosystems. Researchers interested in describing the importance of natural systems to humans frequently begin by generating lists of things normally functioning ecosystems can do. The scope of these lists is not universally constant.

Review of extant attempts to identify the suite of potential functions performed by aquatic ecosystems indicates that the list continues to evolve. The wetlands literature provides one example of this progression. In the 1970s, important wetlands functions included production of plant biomass, provision of habitat, modification of water quality, flood storage, and sediment accumulation (e.g., Wass and Wright, 1969). At present, the list has been expanded considerably and now includes functions in global carbon cycles, maintenance of biodiversity, and global climate control, among others (e.g., Ewel, 2002). There is no reason to believe the list will not continue to evolve as understanding of wetlands and aquatic ecosystems increases.

There have been a number of efforts to develop and suggest a taxonomy for ecosystem functions, and they tend to converge on a generalized categorization suggested by de Groot et al. (2000). These authors argue that the cumulative list of ecosystem functions can be grouped into four primary categories: (1) regulation, (2) habitat, (3) production, and (4) information (see also Table 3-3 below for further information). As described by de Groot and colleagues, regulation functions include those processes affecting gas concentrations, water supply, nutrient cycling, waste assimilation, and population levels. Habitat functions are directly related to provision of suitable living space for an ecosystem's flora and fauna. Production functions include primary (autotrophic) and secondary (heterotrophic) production, as well as generation of genetic material and biochemical substances. Information functions are those that provide an opportunity for cognitive development and, as such, are functions that can be realized only through human interaction.

The committee's review of the literature and attempts to catalog ecosystem functions leads to the conclusion that the absence of a consensus taxonomy is a product of both the complexity of natural systems and the challenge of communicating across multiple disciplines. The committee could find underlying logic in many of the alternative approaches, but no single approach was without complications, and none was intuitively explanatory across disciplines or to all reviewers. For the present, this appears to be the state of the science.

Although a perfect taxonomy for ecosystem functions remains elusive, this may be less important than developing a consensus on an appropriate cumulative list of potential aquatic ecosystem functions. In this regard, de Groot et al. (2000) represent an important iteration in the process of generating a useful checklist to inform aquatic ecosystem valuation exercises. Although the committee found reasons to debate aspects of the proposed listing, the value as stimulus to discussion was clear. Continued work on such compilations will enhance our ability to develop more comprehensive ecosystem valuation scenarios. In the interim, it seems that using a relatively detailed list of ecosystem functions (and goods and services; see more below) like that provided by de Groot et al. (2002) can offer guidance to help ensure some breadth to the assessment of specific ecosystems.

Unfortunately, identification of the particular functions performed by an aquatic ecosystem is only part of the assessment problem. The level at which specific ecosystem functions are performed can also vary significantly, in part because these systems can vary so widely in terms of their physical and biological composition. Thus, production functions can reach extreme levels in eutrophic ponds and estuaries or drop to very low levels in oligotrophic lakes. Climate regulation functions can occur and take on great importance at very high levels in the Great Lakes or be effectively nonexistent in small prairie potholes (wetlands). Thus, while almost all ecosystem functions can be argued to occur at some level in every aquatic ecosystem, the significance of the processes can vary from great to trivial depending on the type of system, its size, and location.

Time can be another important dimension in appropriate assessment of ecosystem function, particularly when economic valuation is the end objective. The rates at which various ecological processes occur will affect their ease of recognition and measurement. For example, habitat functions are arguably easier to identify and measure than carbon sequestration, whereas primary production is easier to assess than generation of genetic material. The frequency with which certain functions are performed can similarly influence recognition and measurement. Production may be a relatively constant or at least seasonal process, while hydroperiod modification may only occur at irregular intervals of years' duration. Finally, the developmental state of the ecosystem will affect its capacity to sustain performance of certain functions. Most aquatic ecosystems change overtime; ponds fill in or dry up, rivers meander and get dammed, and tidal marshes erode. All of these changes alter the capacity of an ecosystem to perform functions over very short to very long time periods.

As a result of the inherent variability in both structure and functions of natural systems, there is no straight forward methodology (let alone a consensus paradigm) for comprehensive assessment of each and every type of aquatic ecosystem. The practical default approach is to work from an evolving list of potential ecosystem functions (e.g., de Groot et al., 2002; MEA, 2003) and evaluate the capacity of the system under consideration to perform each function.

Essential to the process is incorporation of both spatial and temporal considerations in developing the ecosystem assessment.

## Ecosystem Goods and Services

Daily (1997) states that "ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life. They maintain biodiversity and the production of ecosystem goods. . ." Many of the goods and services provided by aquatic ecosystems are intuitive, such as potable water sources, food production, transportation, waste removal, and contributing to landscape aesthetics. To a great extent ecologists are able to catalogue and estimate these kinds of goods and services at both small and large spatial scales. Extending those assessments of goods and services through time is more challenging as ecosystems are constantly changing.

Other, less intuitive, goods and services have been recognized only as knowledge of the global ecosystem has evolved. Some of these include maintenance of biodiversity, and contributing to biogeochemical cycles and global climate. As noted previously, it is likely that the list of potential ecosystem goods and services will continue to evolve.

Reviewers of the subject area have tried to catalog ecosystem goods and services in a variety of ways. Services are sometimes grouped from the perspective of human users into categories such as extractive and nonextractive or consumptive and nonconsumptive. A compilation of some sample lists is included in Table 3-2. Reviewers have also attempted to articulate the link between ecosystem functions and the derived goods and services. One previously noted example of this approach is the de Groot et al. (2002) taxonomy for ecosystem functions, goods, and services shown in Table 3-3.

The state of the science is such that there is no broad consensus on a comprehensive list of potential goods and services derived from aquatic ecosystems. However, there is enough similarity among proposed lists to suggest that full valuation of any particular ecosystem's goods and services must look well beyond the amounts of water, fish, waste assimilation, and recreational use provided to individuals in direct contact with the system. At present, ecologists can quantify many of the more readily accepted goods and services, although methods may vary. It is noteworthy that the international Millennium Ecosystem Assessment (MEA; see also Chapter 2) being coordinated by the United Nations Environment Programme has adopted a taxonomy of ecosystem services drawn from the de Groot et al. (2002) construct (MEA, 2003). After considering a number of alternative schemes for grouping ecosystem services, the approach based on function was selected for use in the MEA. In this particular iteration, services are classified as provisioning, regulating, cultural, or supporting.

TABLE 3-2 Lists of Ecosystem Services

| |
|---|
| Ecosystem Services (Daily, 1997) |
| Purification of air and water |
| Mitigation of floods and droughts |
| Detoxification and decomposition of wastes |
| Generation and renewal of soil and soil fertility |
| Pollination of crops and natural vegetation |
| Control of the vast majority of potential agricultural pests |
| Dispersal of seeds and translocation of nutrients |
| Maintenance of biodiversity, from which humanity has derived key elements of its agricultural, medicinal, and industrial enterprises |
| Protection from the sun's harmful ultraviolet rays |
| Partial stabilization of climate |
| Moderation of temperature extremes and the force of winds and waves |
| Support of diverse human cultures |
| Providing aesthetic beauty and intellectual stimulation that lift the human spirit |
| Services Provided by Rivers, Lakes, Aquifers, and Wetlands (Postel and Carpenter, 1997) |
| Water Supply |
| Drinking, cooking, washing, and other household uses |
| Manufacturing, thermoelectric power generation, and other industrial uses |
| Irrigation of crops, parks, golf courses, etc. |
| Aquaculture |
| Supply of Goods Other Than Water |
| Fish |
| Waterfowl |
| Clams and mussels |
| Pelts |
| Nonextractive or Instream Benefits |
| Flood control |
| Transportation |
| Recreational swimming, boating, etc. |
| Pollution dilution and water quality protection |
| Hydroelectric generation |
| Bird and wildlife habitat |
| Soil fertilization |
| Enhanced property values |
| Nonuser values |
| Wetland Ecosystem Services (Ewel, 2002) |
| Biodiversity: Sustenance of Plant and Animal Life |
| Evolution of unique species |
| Production of harvested wildlife: |
| Water birds, especially waterfowl |
| Fur-bearing mammals (e.g., muskrats) |
| Reptiles (e.g., alligators) |
| Fish and shellfish |
| Production of wildlife for nonexploitative recreation |
| Production of wood and other fibers |
| Water Resources: Provision of Production Inputs |
| Water quality improvements |
| Flood mitigation and abatement |
| Water conservation |
| Global Biogeochemical Cycles: Provision of Existence Values |
| Carbon accumulation |
| Methane production |
| Denitrification |
| Sulfur reduction |
| Ocean Ecosystem Services (Peterson and Lubchenco, 2002) |
| Global materials cycling |
| Transformation, detoxification and sequestration of pollutants and societal wastes |
| Support of the coastal ocean-based recreation, tourism, and retirement industries |
| Coastal land development and valuation, |
| Provision of cultural and future scientific values |

SOURCE: Adapted from Daily (1997); Ewel (2002); Peterson and Lubchenco (2002); Postel and Carpenter (1997).

TABLE 3-3 Functions, Goods, and Services of Natural and Seminatural Ecosystems

| Functions | Ecosystem Processes and Components | Goods and Services |
|---|---|---|
| **Regulation** | Maintenance of essential ecological processes and life support systems | |
| Gas regulation | Role of ecosystems in biogeochemical cycles | Ultraviolet-B protection<br>Maintenance of air quality<br>Influence on climate |
| Climate regulation | Influence of land cover and biologically mediated processes | Maintenance of temperature, precipitation |
| Disturbance prevention | Influence of system structure on dampening environmental disturbance | Storm protection<br>Flood dampening |
| Water regulation | Role of land cover in regulating runoff and river discharge | Drainage and natural irrigation<br>Medium for transport |
| Water supply | Filtering, retention, and storage of fresh water (e.g., in aquifers) | Provision of water for consumptive use |
| Soil retention | Role of vegetation root matrix and soil biota in soil retention | Maintenance of arable land<br>Prevention of damage from erosion and siltation |
| Soil formation | Weathering of rock, accumulation of organic matter | Maintenance of productivity on arable land |
| Nutrient regulation | Role of biota in storage and recycling of nutrients | Maintenance of productive ecosystems |
| Waste treatment | Role of vegetation and biota in removal or breakdown of xenic nutrients and compounds | Pollution control and detoxification |
| Pollination | Role of biota in movement of floral gametes | Pollination of wild plants species |
| Biological control | Population control through trophic-dynamic relations | Control of pests and diseases |
| **Habitat** | Providing habitat (suitable living space) for wild plant and animal species | |
| Refugium | Suitable living space for wild plants and animals | Maintenance of biological and genetic diversity<br>Maintenance of commercially Harvested species |
| Nursery | Suitable reproductive habitat | Hunting; gathering of fish, game, fruit, etc.<br>Aquaculture |
| **Production** | Provision of natural resources | |
| Food | Conversion of solar energy into edible plants and animals | Building and manufacturing<br>Fuel and energy<br>Fodder and fertilizer |
| Raw materials | Conversion of solar energy into biomass for human construction and other uses | Improve crop resistance to pathogens and pests |
| Genetic resources | Genetic material and evolution in wild plants and animals | Drugs and pharmaceuticals<br>Chemical models and tools<br>Test and assay organisms |

TABLE 3-3 Continued

| Functions | Ecosystem Processes and Components | Goods and Services |
|---|---|---|
| Medicinal resources | Variety of (bio)chemical substances in, and other medicinal uses of, natural biota | |
| Ornamental resources | Variety of biota in natural ecosystems with (potential) ornamental use | Resources for fashion, handicraft, worship, decoration, etc. |
| **Information** | Providing opportunities for cognitive development | |
| Aesthetic | Attractive landscape features | Enjoyment of scenery |
| Recreation | Variety in landscapes with (potential) recreational uses | Ecotourism |
| Cultural and artistic | Variety in natural features with cultural and artistic value | Inspiration for creative activities |
| Spiritual and historic | Variety in natural features with spiritual and historic value | Use of nature for religious or historic purposes |
| Science and education | Variety in nature with scientific and educational value | Use of nature for education and research |

SOURCE: Adapted from de Groot et al. (2002).

## ISSUES AFFECTING IDENTIFICATION OF GOODS AND SERVICES

Ecosystems vary in time and space. As ecologists extend their analyses of ecosystem structure and function to include potential goods and services, the uncertainty affecting assessments increases across both time and space. The interaction of ecological and social systems makes extrapolation of observations and prediction of future conditions exceptionally complex (Berkes et al., 2003; Gunderson and Holling, 2002; Gunderson and Pritchard, 2002). The challenges arise from the heterogeneity of systems and values across space which complicates aggregation for assessment at larger scales, and from nonlinear system behavior that confounds forecasting. Recognition of the thresholds of change in both space and time is one of the principal challenges in ecological research.

### Scale

It may be argued that almost all ecosystem functions can be performed by aquatic ecosystems at any scale. Indeed, Limburg et al. (2002) found that scaling rules describing production and delivery of ecosystem services are yet to be formulated and quantified (as noted in the preceding sections). However, there are clearly thresholds in the level of their relative importance. For example, individual wetlands in a watershed may each have the capacity to slow the flow of waters moving through them, but this function becomes important only when there are a sufficient number of wetlands in a watershed to significantly alter the flow of floodwaters downstream.

The complication in assessment of ecosystem goods and services arises because the scale at which functions become important is not always the same. Continuing with the watershed example above, each wetland may have the capacity to accrete organic matter, sequestering carbon. However, the significance of this function for carbon cycles may not be realized at any

scale less than all of the nation's wetlands. Alternatively, the provision of suitable habitat for a rare plant may be regionally significant at the scale of a single wetland.
Some generalizations regarding recognition of ecosystem services across scales may be possible (see Table 3-4 for one example). The problem is recognition of the thresholds at resolution sufficient to inform management and policy decisions. Knowing precisely the scale at which services can be realized is a practical challenge. Success in identification of these scale thresholds would increase opportunities for accurate recognition and appropriate economic valuation of ecosystem services.

Another challenge in valuing ecosystem services across scales arises in attempts to aggregate such information. The complex nature of ecosystems means that many interrelationships and feedback loops may operate at scales above the level of individual service assessment. Protection of wetlands important as habitat for migrating waterfowl may be undermined by loss of wetlands at other critical points on the flyway. Restoration of wetlands as nursery grounds for fish along the Louisiana coast may be less successful if nutrient pollution in the Mississippi River degrades open water habitat for the adult populations. The implication is that aggregation of service values to larger scales or composite system evaluations will almost axiomatically misrepresent the processes at the target scale. This is a particularly difficult problem since it is assumed to exist and yet can be managed only by comprehensive knowledge of the system under study.

The uncertainties associated with consideration of scale in assessment of ecosystem goods and services will only be resolved by continuing investigation of natural systems. At present the practical solution is upfront recognition of the potential for aggregation errors and careful framing of the assessment question. Explicit identification of the ecosystem goods and services being evaluated, careful definition of the scale at which those services are generally realized, and comparison to the scale of the assessment being undertaken can at least bound the valuation process and inform subsequent decisions.

TABLE 3-4 Examples of the Generation of Ecosystem Services at Different Scales for Aquatic Ecosystems

| Time or Space Scale (day) (meters) | Aquatic Ecosystem | Example of Ecosystem Service | Scale at Which Service is Valued |
|---|---|---|---|
| $10^{-6}$ to $10^{-5}$ | Bacteria | Nutrient uptake and production of organic matter | Local/regional |
| $10^{-3}$ to $10^{-1}$ | Plankton | Trophic transfer of energy and nutrients | Local/regional |
| $10^{0}$ to $10^{1}$ | Water column and/or sediments, small streams | Provision of habitat | Local |
| $10^{2}$ to $10^{4}$ | Lakes, rivers, bays | Fish and plant production | Local/regional |
| $\geq 10^{5}$ | Ocean basins, major rivers, and lakes | Nutrient regulation, $CO_2$ regulation | Global |

SOURCE: Adapted from Limburg et al. (2002).

## System Dynamics

Natural systems are increasingly understood as dynamic constructs that may exist in a number of alternate states (also referred to as "regimes" or "domains of ecological attraction" depending on the terminology being used). A system may move, or "flip," from one state to another if it passes a threshold of some controlling variable. The transition to an alternate state may be rapid or gradual, and may or may not reflect a change in the trajectory of the system. The concept of alternative states with boundary thresholds is used to explain the nonlinear behavior of natural systems. Indeed, examples of thresholds and regime shifts in aquatic ecosystems have been a significant part of the evolving understanding of nonlinear ecosystem behavior (Muradian, 2001; Scheffer and Carpenter, 2003; Scheffer et al., 2001; Walker and Meyers, 2004).

Many ecosystems can persist in a particular state or regime for some time because they exhibit resistance or resilience. Resistance is measured by the capacity to withstand disturbance without significant change, while resilience is indicated by the capacity to return to the original state after perturbation toward an alternate state. Resilience was originally described by Holling (1978) and persists as an important concept in the analysis of social-ecological system dynamics today (Walker and Myers, 2004; Walker et al. 2004).

The nonlinear system behavior that emerges in response to thresholds and regime shifts can be problematic for assessment of ecosystem services. Recognition of the points at which alternative behavior will emerge is difficult in many systems. (See Figure 3-1 for a conceptual representation of the nonlinear ecosystem response to stress.) As noted by Chavas (2000) ". . . ecosystem dynamics can be highly nonlinear, meaning that knowing the path of a system in some particular situation may not tell us much about its behavior under alternative scenarios."

An example of this type of behavior can be found in the waste assimilation and transport services of lakes, rivers, and estuaries. Increased nutrient loads in an aquatic ecosystem may simply increase productivity of the resident biota up to the point of harmful eutrophication. At that point, the high levels of primary production overwhelm secondary production and decomposition processes, resulting in excessive accumulation of organic matter, depletion of oxygen in the water column, and a change in the trophic structure. The change can represent a new and undesirable condition that may persist even if nutrient loads are reduced (see Carpenter, 2003; Carpenter et al., 1998). From the perspective of ecosystem service assessment, waste assimilation may still be occurring, but habitat services, recreational services and maintenance of biodiversity may all be significantly changed. The point at which this abrupt shift in services occurs may be controversial and unpredictable.

In some circumstances the abrupt shift, or flip to an alternate regime in state may be part of a hysteretic system behavior. In this case the stress threshold that generated the response may be significantly higher than the stress threshold that will allow a recovery. This type of response can be found in many dense and highly productive aquatic communities, such as seagrass beds (Batuik et al., 2000). Often these communities can tolerate significant levels of physical stress simply because there are a sufficient number of individuals to moderate physical conditions inside the community and enough reproductive potential to offset the continual losses. When the physical stresses surpass a community's capacity to withstand them, reestablishment can often succeed only in conditions significantly less stressful than the robust community could tolerate (Molles, 2002). In essence, the recovery threshold differs from the impact threshold such that the state of the system will lag in response to changes in controlling forces.

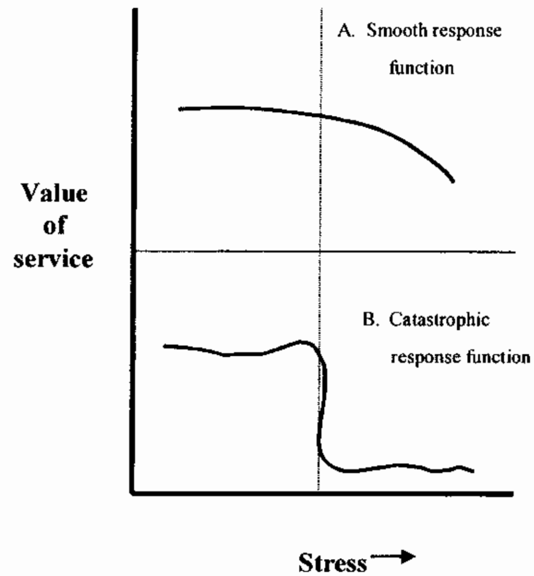


FIGURE 3-1 Value responses to stress under marginal (well-behaved dynamics) and nonmarginal (nonlinear, threshold dynamics) system behaviors. SOURCE: Reprinted, with permission, from Limburg et al. (2002). © 2000 by Elsevier.

Cascading effects are another example of ecosystem dynamics that can be difficult to predict (Molles, 2002). Harvest of top-level predators can result in increases in lower-level predators, decreases in herbivore prey, and resultant changes in vegetation. Alterations in river flows can change the timing of nutrient introductions to downstream waterbodies, resulting in modified phytoplankton and zooplankton communities, and culminating in shifts in habitat quality for higher-trophic-level fish communities.

There is considerable ongoing research to define thresholds and develop indicators of system condition that will assess proximity of thresholds. While understanding of these system dynamics continues to expand, this knowledge can inform assessment of ecosystem functions only if the assessment occurs at appropriate spatial and temporal scales, and appropriate spatial and temporal scales can be identified only if the dynamics are already understood. In the face of this apparent conundrum the practical solution to the need to complete an assessment of ecosystem function and/or provision of services is to proceed with caution. Observations of a system's behavior through time are an obvious first step, but such monitoring data can only confirm the existence of nonlinear behavior, not prove its absence. Simply considering the possibilities for threshold responses may be adequate to inform some assessments, and is certainly preferable to ignoring the issue.

## Intrinsic Values

Many people believe that ecosystems have value quite apart from any human interest in explicit goods or services (see Chapter 2 for further information). The fact that ecosystems exhibit emergent behaviors and operate to sustain themselves is sufficient to argue that they have value to their components. Although comprehending this intrinsic value does not trouble most individuals, assessing it is problematic. Farber et al. (2002) state, "As humans are only one of many species in an ecosystem, the values they place on ecosystem functions, structures and processes may differ significantly from the values of those ecosystem characteristics to species or the maintenance (health) of the ecosystem itself."

### Incomplete Knowledge

Comprehensive valuation of aquatic ecosystems should be viewed as a practical improbability. The assumption that our knowledge is imperfect is at the root of the concern for aggregation of assessments to larger scales and composite valuation of whole ecosystems. As a consequence, unforeseen behaviors and services are anticipated, and valuations are automatically caveated with concern for the state of the science. This does not imply no ecosystem valuation can be accomplished, simply that comprehensive valuation should not be presumed. Many decisions using economic or other valuation techniques can be made without a comprehensive assessment of ecosystem goods and services

An example of how the state of our understanding can impact the capacity to value an ecosystem service involves the relationship between biodiversity and aquatic ecosystem functions. In efforts to identify ecosystem services, researchers typically acknowledge the importance of habitat functions for maintenance of biodiversity. For some time, high biodiversity was assumed to confer some inherent resistance and/or resilience to a system, allowing it to sustain performance of other valued services in the face of disturbance. However, researchers are not of a single mind about the nature of the relationship between biodiversity and ecosystem functioning (e.g., Duarte, 2000; Ghilarov, 2000; Hulot et al., 2000; Schwartz et al., 2000; Ulanowicz, 1996). It can be difficult, if not impossible, therefore to accurately assess the importance of any particular ecosystem's contribution to maintenance of biodiversity, or conversely the role of biodiversity in the functioning of the ecosystem.

Another area in which a lack of comprehensive knowledge limits full recognition of services provided by aquatic ecosystems is the continual growth in the number of ways humans can use aquatic resources. The continually expanding lists of medicinal and industrial products found in aquatic ecosystems provide obvious examples, while the evolving number of aquatic recreational activities is another. The point is that the list of services is not determined entirely by the suite of natural functions in aquatic ecosystems, but also by human ingenuity in deriving benefits.

## SUMMARY: CONCLUSIONS AND RECOMMENDATIONS

In review and discussion of the state of the science in the identification of aquatic ecosystem functions and their linkage to goods and services, the committee arrived at several specific conclusions:

- Ecologists understand the uncertainties in ecosystem analysis and accept them as inherent caveats in all discussions of system performance.

- As the committee pursued its charge, the problems of developing an interdisciplinary terminology and/or a universally applicable protocol for valuing aquatic ecosystems were illuminated, but ultimately identified as unnecessary objectives.
- From an ecological perspective, the value of specific ecosystem functions/services is entirely relative. The spatial and temporal scales of analysis are critical determinants of potential value.
- Potentially useful classification and inventories of aquatic ecosystems as well as their functional condition exist at both regional and national levels, though the relevance of these classification and inventory systems to assessing and valuing aquatic ecosystems is not always clear.
- Ecologists have qualitatively described the structure and function of most types of aquatic ecosystems. However, the complexity of ecosystems remains a barrier to quantification of these features, particularly their interrelationships.
- General concepts regarding the linkages between ecosystem function and services have been developed. Although precise quantification of these relationships remains elusive, the general concepts seem to offer sufficient guidance for valuation to proceed with careful attention to the limitations of any ecosystem assessment.
- Many, but not all, of the goods and services provided by aquatic ecosystems are recognized by both ecologists and economists. These goods and services can be classified according to their spatial and temporal importance.
- Complex ecosystem dynamics and incomplete knowledge of ecosystems will have to be resolved before comprehensive valuation of ecosystems is tractable, but comprehensive ecosystem valuation is not generally essential to inform many management decisions.
- Further integration of the sciences of economics and ecology at both intellectual and practical scales will improve ecologists' ability to provide useful information for assessing and valuing aquatic ecosystems.

There remains a significant amount of research and work to be done in the ongoing effort to codify the linkage between ecosystem structure and function and the provision of goods and services for subsequent valuation. The complexity, variability, and dynamic nature of aquatic ecosystems make it likely that a comprehensive identification of all functions and derived services may never be achieved. Nevertheless, comprehensive information is not generally necessary to inform management decisions. Despite this unresolved state, future ecosystem valuation efforts can be improved through use of several general guidelines and research conducted in the following areas:

- Aquatic ecosystems generally have some capacity to provide consumable resources (e.g., water, food); habitat for plants and animals; regulation of the environment (e.g., hydrologic cycles, nutrient cycles, climate, waste accumulation); and support for nonconsumptive uses (e.g., recreation, aesthetics, research). Considerable work remains to be done in documentation of the potential that various aquatic ecosystems have for contribution in each of these broad areas.
- Delivery of ecosystem goods and services occurs in both space and time. Local and short-term services may be most easily observed and documented, but the less intuitive accumulation of services over larger areas and time intervals may also be significant. Alternatively, services that are significant only when performed over large areas or long time

intervals may be beyond the capacity of some ecosystems. Investigation of the spatial and temporal thresholds of significance for various ecosystem services is necessary to inform valuation efforts.

- Natural systems are dynamic and frequently exhibit nonlinear behavior. For this reason, caution should be used in extrapolation of measurements in both space and time. Although it is not possible to avoid all mistakes in extrapolation, the uncertainty warrants explicit acknowledgment. Methods are needed to assess and articulate this uncertainty as part of system valuations.

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# 4
# Methods of Nonmarket Valuation

## INTRODUCTION

This chapter outlines the major methods that are currently available for estimating economic (monetary) values for aquatic and related terrestrial ecosystem services. Within the chapter is a review of the economic approach to valuation, which is based on a total economic value framework. In addition to presenting the valuation approaches, the chapter discusses the applicability of each method to valuing ecosystem services. It is important to note that the chapter does not instruct the reader on how to apply each of the methods, but rather provides a rich listing of references that can be used to develop a greater understanding of any of the methods. Based on this review, the chapter includes a summary of its conclusions and recommendations.

The substance of this chapter differs from the various books and chapters that provide overviews of nonmarket valuation methods (e.g., Braden and Kolstad, 1991; Champ et al., 2003; Herriges and Kling, 1999; Mäler and Vincent, 2003; Mitchell and Carson, 1989; Ward and Beal, 2000) because these prior contributions were designed to summarize the state of the art in the literature or to teach novices how to apply the various methods. This chapter also differs from government reports that provide guidance for implementing nonmarket valuation methods (NOAA, 1993; EPA, 2000a). The purpose of this chapter is to carefully lay out the basic valuation approaches and explain their linkages to valuing aquatic ecosystems. This is done within the context of the committees' implicit objective (see Box ES-1) of assessing the literature in order to facilitate original studies that will develop a closer link between aquatic ecosystem functions, services, and value estimates.

## ECONOMIC APPROACH TO VALUATION

### Economic Valuation Concepts

As discussed in Chapter 2, the concept of economic valuation adopted in this report is very broad. That is, the committee is concerned with how to estimate the impacts of changes in ecosystem services on the welfare, or utility (satisfaction or enjoyment), of individuals. If ecosystem changes result in individuals feeling "worse off," then one would like to have some measure of the loss of economic value to these individuals. Alternatively, if the changes make people "better off," one would like to estimate the resulting value gain.

The basic concepts that economists use to measure such gains and losses are economic values measured as a *monetary payment* or a *monetary compensation.* The essence of this approach is to estimate values as subtractions from or additions to income that leave people equally economically satisfied with or without a change in the services provided by an aquatic ecosystem. For example, suppose a lake was contaminated with polychlorinated biphenyls (PCBs) discharged by a nearby factory. In such a case, the logical valuation concept is an estimate of the monetary compensation that is required to bring the affected people back to the same level of satisfaction they enjoyed prior to the contamination event. Such a measure of value, when aggregated over all affected people, could be used to assign a damage payment to the factory responsible for the pollution. Funds collected from the polluter would not typically be paid directly to the affected people, but would be used for restoration projects that would return services to the lake.

Another type of application would be a project to enhance a freshwater wetland to improve sportfishing opportunities. In this example, one group of people consists of the direct beneficiaries, people who fish recreationally. Valuation would be used to estimate the "maximum" that anglers would pay for this improvement in fishing. Although no money would actually be collected from the anglers, each angler's expression of his or her maximum willingness to pay represents how much the angler is prepared to compensate the rest of society for the increased individual enjoyment gained from the improved recreational fishing. Maximum willingness to pay is aggregated for all anglers who benefit to determine whether the benefits of the wetland project exceed the costs, which facilitates an assessment of whether public funds should be spent on the project.

These two examples provided several insights:

1. Values arise from the preferences of individual people; thus, values are estimated for individuals or households and then aggregated to obtain the values that society places on changes in aquatic ecosystems.
2. Valuation methods are used to estimate the gains or losses that people may experience as a result of changes in aquatic ecosystems in order to inform policy discussions and decisions.
3. Different types of changes in aquatic ecosystems affect different groups of people, which, as discussed in more detail below, may influence the choice of valuation methods used.
4. There are two basic concepts of value (noted elsewhere in this report), willingness to accept (WTA) (compensation) and willingness to pay (WTP).[1]

Whether WTA or WTP is conceptually the appropriate measure of value for changes in aquatic ecosystems depends on the presumed endowment of property rights. In the case of PCB contamination, the presumed property right of society was to a lake that is free of PCBs. This implies that the conceptually appropriate value measure that would restore people to their original level of satisfaction is WTA compensation. In contrast, in the freshwater wetland restoration example, the presumed property right is in the existing fishing conditions and the appropriate value measure is WTP to obtain the improvement in fishing conditions. Unfortunately, economists have had difficulty in measuring WTA (Boyce et al., 1992; Brown and Gregory, 1999; Coursey et al., 1987; Hanemann, 1991) and most empirical work for policy applications involve measures of WTP. This issue arises for a variety of reasons, such as survey respondents not being familiar with WTA questions and because most respondents have incomplete knowledge of relative prices. Thus, most of the following discussion focuses on the use of valuation methods to estimate WTP.

[1] For further discussion of measurements of WTP and WTA, see Chapter 2.

## Why Valuation Is Required

Chapter 2 discusses the importance of economic valuation as input into decision-making and, in particular, for aiding the assessment of policy choices or trade-offs concerning various management options for aquatic ecosystems. As Chapter 3 has illustrated, given the complex structure and functioning of aquatic and related terrestrial ecosystems, these systems often yield a vast array of continually changing goods and services. The quality and quantity of these services are in turn affected by changes to ecosystem structure and functioning. Thus, alternative policy and management options can have profoundly different implications for the supply of aquatic ecosystem services, and it is the task of economic valuation to provide estimates to decision-makers of the aggregate value of gains or losses arising from each policy alternative.

Valuation is especially important because many services provided by aquatic ecosystems have attributes of public goods. Public goods are are nonrival and nonexcludable in consumption, which prevents markets from efficiently operating to allocate the services. An example would be wetland filtration of groundwater. As long as the quantity of groundwater is not limiting, everyone who has a well in the area can enjoy the benefits of unlimited potable groundwater. However, in the absence of any market for the provision of water through wetland filtration, there is no observed price to reveal how much each household or individual is willing to pay for the benefits of this service. Although everyone is free to use the aquifer, no one is responsible for protecting the aquifer from contamination. This is not an action that could be undertaken by a company and provided for a fee (price) because no individual has ownership of neither the wetland filtration process nor the aquifer. Nonmarket values can be estimated to reveal whether the benefits of collective action—perhaps through a state environmental agency or the U.S. Environmental Protection Agency (EPA)—exceed the cost of the proposed actions to protect the wetland, and consequently the wetland filtration process and the quality of the waster in the aquifer for drinking purposes.

It is also the case that some aquatic ecosystem services indirectly contribute to other services that are provided through a market, but the value of this ecological service itself is not traded or exchanged in a market. For example, an estuarine marshland may provide an important "input" into a commercial coastal fishery by serving as the breeding ground and nursery habitat for fry (juvenile fish). Although disruption or conversion of marshland may affect the biological productivity of the marsh, and thus its commercial fishery, a market does not exist for the commercial fishery to pay to maintain the habitat service of the marshland. The problem is also one of transaction costs. It is costly for participants in the commercial fishery to get together to negotiate with owners of marshland and there may be many owners of marshland for which protection agreements must be sought. Estimation of the implicit (nonmarket) value to the fishery of marsh habitat can be used to understand whether laws and rules to protect the breeding and nursery functions of the marsh.

Aquatic ecosystem services that do not have market prices are excluded from explicit consideration in cost-benefit analyses and other economic assessments, and are therefore likely to not get full consideration in policy decisions. As noted in Chapter 2, Executive Order 13258, which supersedes Executive Orders 12866[2]; and EO 12291[3] requires government agencies to demonstrate that the benefits of regulations outweigh the costs. (All of the benefit-cost discussion occurs in Executive Order 12866 and federal agencies still reference this order.) This

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[2] Executive Order 12866. October 4, 1993. Federal Register 58 (190).
[3] Executive Order 12291. February 19, 1981. Federal Register 46(33).

mandate is followed by the U.S. Environmental Protection Agency (EPA, 2000a) *Guidelines for Preparing Economic Analyses*, which emphasizes the importance of valuation to decision-making on the environment. Thus, if monetary values for ecosystem services are not estimated, many of the major benefits of aquatic ecosystems will be excluded in benefit-cost computations. The likely outcome of such an omission would be too little protection for aquatic ecosystems, and in consequence the services that people directly and indirectly enjoy would be under-supplied. Valuation, therefore, can help to ensure that ecosystem services that are not traded in markets and do not have market prices receive explicit treatment in economic assessments. The goal is not to create values for aquatic ecosystems. Rather, the purpose of valuation is to formally estimate the "nonmarket" values that people already hold with respect to aquatic ecosystems. Such information on nonmarket values will in turn assist in assessments of whether to protect certain types of aquatic ecosystems, to enhance the provision of selected ecosystem services, and to restore damaged ecosystems.

Finally, economic values are often used in litigation involving damage to aquatic ecosystems from pollution or other human actions. For evidence to be credible, including ecosystem modeling and economic values, it must pass a Daubert test,[4] the essential points of which are whether the following apply:

- the theories and techniques employed by the scientific expert have been *tested*;
- they have been subjected to *peer review* and *publication*;
- the techniques employed by the expert have a *known error rate*;
- they are subject to *standards* governing their application; and
- the theories and techniques employed by the expert enjoy widespread acceptance.

All of the nonmarket valuation methods discussed in this chapter meet these conditions in general. A key issue, and thus theme of this chapter is which of the methods are applicable to valuing the services of aquatic and related terrestrial ecosystems and under what conditions and circumstances? Issues raised throughout this chapter suggest areas in need of original research between ecologists and economists that will ultimately provide better aquatic ecosystem value estimates to support policy evaluations and decision-making that are defensible.

## The Total Economic Value Framework

As discussed in Chapter 2, the total economic value (TEV) framework is based on the presumption that individuals can hold multiple values for ecosystems and is developed for categorizing these various multiple benefits. Although any taxonomy of values is somewhat arbitrary and may differ from one use to another, the TEV framework is necessary to ensure that some components of value are not omitted in empirical analyses and that double counting of values does not occur when multiple valuation methods are employed. For example, Table 3-2 presents several categorizations of ecosystem services. In any empirical application it is necessary to map these services to how they affect humans and then select an appropriate valuation method. This chapter presents information that helps with the selection of a valuation approach, while Chapter 5 discusses the mapping of changes in ecosystem to effects upon

[4] For further information about Daubert test, see *http://www.daubertontheweb.com/Chapter_2.htm.*

humans through a series of case studies. The TEV approach presents a road map that facilitates this mapping of ecosystem services to effects and the selection of valuation methods.

### Valuation Under Uncertainty

Estimation of use and nonuse values (see Chapter 2 for a detailed discussion of use and nonuse values; see also Table 2-1) is often associated with uncertainty. For example, current efforts to restore portions of the Florida Everglades (see also Chapter 5 and Box 3-6) do not imply that the original services of this wetland area can be restored with certainty. It is also impossible to predict with certainty the changes in service provided by aquatic ecosystems due to global warming. These situations are not unique when aquatic ecosystem services are valued. In addition, individuals may be uncertain about their future demand for the services provided by restoration of the Everglades or the services affected by global warming. For example, someone living in New York may be unsure if they will ever visit the Everglades, which affects how they might value the improvements in opportunities to watch birds in the Everglades. Someone who lives in the Rocky Mountain states may be unsure about whether they will ever visit the Outer Banks in North Carolina, which affects the value they place on losing this coastal area to erosion.

These uncertainties can affect the estimation of use and nonuse values from an ex ante ("beforehand") perspective. The economist's concept of TEV for ex ante valuation under uncertainty, from either the supply or the demand side, is *option price* (Bishop, 1983; Freeman, 1985; Larson and Flacco, 1992; Smith, 1983; Weisbrod, 1964).[5] The notion of option price follows that of TEV, whereas option value is simply the concept of TEV when uncertainty is present and includes all use and nonuse values an individual holds for a change in an aquatic ecosystem. Option price is the amount of money that an individual will pay or must be compensated to be indifferent between the status quo condition of the ecosystem and the new, proposed condition. Option prices can be estimated for removing the uncertainty or for simply changing probabilities; reducing the probability of an uncertain event (beach erosion) or increasing the probability of a desirable event (e.g., increased quality of bird watching). Option prices are also estimated for conditions where probabilities do not change, but the quantity or quality associated with a probability changes.

The following section of the chapter focuses on the micro-sense of uncertainty in the estimation of individual, or perhaps household, values, whereas Chapter 6 takes a broader perspective of uncertainty that includes how values estimated in the presence of uncertainty are used to inform policy decisions. The discussion in Chapter 6 includes concepts such as "quasi-option value" and its relationship to option values.

## CLASSIFICATION OF VALUATION APPROACHES

Since economists often employ a variety of methods to estimate the various use and nonuse values depicted in Table 2-1, another common classification is by *measurement*

[5] Another component of value, *option value*, is commonly referred to as a nonuse value in the literature (see Chapter 6 for further information). Option value arises from the difference between valuation under conditions of certainty and uncertainty and is a numerical calculation, not a value held by people. The literature cited above makes this distinction and does not mistakenly include option value as a component of TEV.

*approaches*. As shown in Table 4-1, this type of categorization is usually organized according to two criteria:

1. whether the valuation method is to be based on *observed* economic behavior, from which individual preferences can be inferred, or whether the valuation method is to be based on responses to survey questions that reveal *stated preferences* by individuals, and
2. whether monetary estimates of values are observed directly or inferred through some indirect method of data analysis.

Because of the public good nature of many of the services described previously, market prices do not exist. Simulated markets are typically used as a benchmark to judge the validity of value estimates derived from indirect methods, but simulated markets are rarely used to develop policy-relevant estimates of value. The open-ended format is not commonly used in contingent valuation studies due to problems with zero bids and protest responses (Bateman et al., 2002; Boyle, 2003). Indirect methods are the most commonly used approaches to valuing aquatic ecosystem services, and the discussion below focuses on these approaches.

## Household Production Function Methods

Household production function (HPF) approaches involve modeling consumer behavior, based on the assumption of a substitutional or complementary relationship between an ecosystem service and one or more marketed commodities. The combination of the environmental service and the marketed commodities, through a household production process, results in the "production" of a utility-yielding good or service (Bockstael and McConnell, 1983; Freeman, 1993a; Mäler, 1974; Smith, 1991, 1997). Examples of these approaches include time allocation models for collecting water, travel-cost methods for estimating the demand for visits to a recreation site, averting behavior models that are frequently used to measure the health impacts of pollution, and hedonic property value or wage models.

TABLE 4-1 Classification of Valuation Approaches

| | Revealed Preferences | Stated Preferences |
|---|---|---|
| Direct | Competitive market prices<br>Simulated market prices | Contingent valuation, open-ended response format |
| Indirect | Household production function models<br>Time allocation<br>Random utility and travel cost<br>Averting behavior<br>Hedonics<br>Production function models<br>Referendum votes | Contingent valuation, discrete-choice and interval response formats<br>Contingent behavior<br>Conjoint analysis (attribute based) |

SOURCE: Adapted from Freeman (1993a).

The inspiration for HPF approaches is the "full income" framework for determining household resource allocation and consumption decisions as developed by Becker (1965), although the HPF model can be applied to a valuation problem without assuming a single, "full income" constraint. The HPF provides a framework for examining interactions between purchases of marketed goods and the availability of nonmarket environmental services, which are combined by the household through a set of technical relationships to "produce" a utility-yielding final good or service. For example, in the presence of contaminated drinking water a household would be expected to invest time and purchased inputs (e.g., an averting technology, bottled water, etc.) to provide a desired service, namely potable water. This is the essence of the averting behavior approach, and in the above example the household is attempting to avoid exposure to a degraded drinking water system.

Appendix B, using travel-cost models, averting behavior approaches, and hedonic price methods, illustrates that the assumptions underlying the "household production function" will vary depending on the environmental problem and the valuation approach. Nevertheless, the common theme in all applications of the HPF approach is the derivation of derived demand for the environmental asset in question. Thus, information on the value of environmental quality can be extracted from information on the household's purchases of marketed goods. The following section illustrates the HPF framework with three examples applied to aquatic ecosystems: (1) random utility or travel-cost models, (2) averting behavior models, and (3) hedonic models.

### *Random Utility and Travel-Cost Models*

The modern variants of travel-cost models are known as random utility models (RUMs). Random utility models arise from the empirical assumption that people know their preferences (utility) with certainty, but there are elements of these preferences that are not accessible to the empirical observer (Herriges and Kling, 1999; Parsons, 2003a). Thus, parameters of peoples' preferences can be recovered statistically up to a random error component. This econometric approach is used to estimate modern travel-cost models. The most common application of this modeling framework has been valuing recreational fishing in freshwater lakes and rivers and marine waters.

Travel-cost studies attempt to infer nonmarket values of ecological services by using the travel and time costs that an individual incurs to visit a recreation site (Bockstael, 1995). Out-of-pocket travel costs and the opportunity cost travel time are used as the implicit price of visiting a site, perhaps a lake to fish or swim. Traditional travel-cost studies utilized the implicit price of travel and the number of times each individual in a sample visited a site to estimate the demand for visits to the site. If the site is a lake and the recreation activity is fishing, this approach yields an in situ value for fishing at the site, only part of which is attributable to the aquatic ecosystem services. The values of ecosystem services are fixed for any given lake at a specific point in time and cannot be identified statistically.

In the case of qualitative differences in the ecological attributes and thus the recreational potential of different sites, random utility models have been employed to value changes in the desirable ecological characteristics that make each site attractive for recreation. The advantage of the RUM approach over traditional travel-cost studies is that, by assuming each recreational site option is mutually exclusive, it is possible to determine how ecological characteristics or attributes of each site affect the decision of an individual to select one particular site for recreation. Thus, the RUM approach is uniquely designed to estimate values for attributes of recreation sites, which for fishing include the quantity and quality of the aquatic ecosystem services. The RUM approach looks at peoples' choices of recreation sites among the menu of available sites and determines the implied values people hold for site attributes by making choices between sites that vary in terms of the cost of visiting the sites and their component attributes, which include aquatic ecosystem characteristics. All other factors being equal, the basic premise of the travel-cost approach is that people will choose the site with the lowest travel cost. When two sites have equal travel costs, people will choose the site with higher quality. If one site has more desirable species of fish, say native trout, then that site will be chosen. Alternatively, if one site has degraded water quality that results in a fish consumption advisory, that site would not be chosen. RUMs use information on these revealed choices to estimate the values people place on aquatic ecosystem services that support recreational opportunities. That is, people will travel further to improve the quality of their visit to an aquatic ecosystem. This behavior allows the empirical investigator to infer the value that individuals place on an improvement or degradation in an aquatic ecosystem.

Another aspect of RUMs is that they can be designed to allow the number of participants to increase (or decrease) as an ecosystem is enhanced (or diminished). The individual actually faces three choices: (1) whether to participate in an activity (e.g., sportfishing), (2) where to go fishing on any particular occasion, and (3) how often to participate in fishing. This is important because both the average value per visit per person, the number of visits an individual makes, and the number of affected people determine aggregate, societal values. While travel-cost models and their modern RUM variants are based on the conceptual framework of household production technology, the production is generally assumed to be undertaken on an individual basis and values are estimated for individuals, not households.

A common concern of human interactions with ecosystems is the potential for the extinction of species through pollution, destruction of habitat, and overuse by humans. All of these factors come into play for the Atlantic salmon in Maine rivers. The rivers in Maine have been heavily dammed to provide hydroelectric power, which diminishes and destroys salmon habitat. There is a long history of pollution by the timber industry and communities, which diminishes water quality for salmon. There has also been substantial fishing pressure, both commercial and recreational, on Atlantic salmon. Morey et al. (1993) employed a RUM to estimate the values that recreational anglers place on salmon fishing. They used a model in which anglers choose among eight salmon fishing rivers in Maine and the Canadian provinces of New Brunswick, Nova Scotia, and Quebec. This area includes all of the major salmon fishing rivers in the northeastern United States and eastern Canada readily accessible to U.S. citizens by car. The authors estimated values for a scenario that asked what the loss per angler would be if salmon numbers fell to the point that anglers are not longer able to fish the Penobscot River in Maine. The Penobscot River is the major salmon fishing river in Maine and this scenario would estimate losses if the river was closed to fishing, for example, because Atlantic salmon in the Penobscot River were listed as endangered so that fishing would be prohibited. The annual loss per angler of not being able to fish the Penobscot, but still being able to fish one of the other seven sites in the model was about \$800. They also estimated a model that asked what would happen if restoration of salmon to the Penobscot River increased the salmon population so that catch rates doubled. The annual benefit per angler was about \$650 per year. The first scenario estimates the value for loss of an ecosystem service, and no specific information from ecologists was needed to estimate this value. The second scenario estimates a value from an improvement in ecosystem services. To develop the estimate for the latter scenario, Morey et al. (1993) included angler catch rates in their model and sportfishing as an indicator of the quality of the ecosystem services enjoyed by people. Two important considerations arise here. First, in order to simulate a doubling of catch rates on the Penobscot River it is necessary for other fishing sites to have catch rates that approximate a doubling of the catch rate for the Penobscot. This means that value predictions are within the range of quality over which anglers have exhibited revealed behavior. This provides observations of revealed choice for this change in quality. Second, absent from the model was a link between salmon populations in the Penobscot River and catch rates. To make the latter scenario realistic for policy analyses it would be necessary to model the relationship between catch rates and population to know what population of salmon is necessary in the Penobscot River to support this doubling of service. Although there is nothing technically wrong with the value estimates reported, there is no direct ecosystem link to indicate how a biological intervention would affect catch rate and the subsequent catch rate could be used to estimate a policy-relevant value. At present, the values reported are simply illustrative. This also opens the question of what has to be undertaken from an ecological perspective to enhance the population of Atlantic salmon in the river.

Another interesting RUM application is also a sportfishing study. In this study, researchers looked at the effect of fish consumption advisories on choices of sportfishing site (Jakus et al., 1997; see also Jakus et al., 1998). Here the ecosystem service is the effect on human health from consumption of fish. However, this service has been diminished by pollution at some sites, which has been signaled to anglers through consumption advisories (i.e., official warnings not to fish). -This study considered fishing on 22 reservoirs in Tennessee, 6 of which had consumption advisories against fishing. Only reservoirs that were within 200 miles of an angler's residence were considered possible fishing sites in the model. Jakus and colleagues found that removing fish consumption advisories from the two reservoirs within 200 miles of residents of central Tennessee had a value of $22 per angler per year. Likewise, removing the advisories from six reservoirs within 200 miles of residents of east Tennessee would have a value of $47 per angler per year. These are estimates of the damages from pollution as signaled by fish consumption advisories. From a policy perspective, to compute aggregate losses it is necessary to know whether ecological restoration will allow removal of the advisories and when this might occur. Thus, the losses of $22 and $47 per angler per year will continue to accumulate each year that the advisories remain in place.

Other studies that have used RUMs to estimate values for aquatic ecosystem services include the following:

- effects of river and reservoir water levels on recreation in the Columbia River basin (Cameron et al., 1996);
- fishing in the Great Lakes (Phaneuf et al., 1998);
- fishing in freshwater lakes (Montgomery and Needleman, 1997);
- river fishing (Morey and Waldman, 1998);
- fishing and viewing wildlife in wetlands (Creel and Loomis, 1992);
- fishing in coastal estuaries (Greene et al., 1997);
- swimming in lakes (Needleman, and Kealy, 1995);
- beach use (Haab and Hicks, 1997);
- boating on lakes (Siderelis et al., 1995); and
- effects of climate change on fishing (Pendleton and Mendlesohn, 1998).

The largest majority of RUMs have valued recreational fishing in lakes (Parsons, 2003b), but as the above examples indicate, there have been applications to other types of aquatic ecosystems and services. Even some terrestrial applications may have relevance to aquatic ecosystem services valuation. For example, one of the early RUM applications was to downhill skiing (Morey, 1981). As ski areas continue to draw more surface water to make snow, there are likely to be increasing impacts on nearby aquatic ecosystems. Thus, policies that affect how much surface water can be used to make snow will have an effect on the value people place on downhill skiing.

The most common use of RUMs is to estimate the in situ value of visiting a recreational site that is related to an aquatic ecosystem. The typical effects of ecosystem services valued in RUMs are changes in fish catch rates, the presence of fish consumption advisories, and degradation of surface waters due to eutrophication from nonpoint pollution. Rarely are other dimensions of ecological services of aquatic ecosystems valued. The key element of applications of RUMs to aquatic ecosystems is that there must be a service that affects the sites people choose to visit. This could include fish catch rates, fish consumption advisories, or waters levels, as demonstrated in the studies cited above. This is by no means an exhaustive list of services, just the obvious services that have been commonly used in developing RUMs.

RUMs have typically been applied to single-day recreation trips and have not examined multiple-day trips. The reason for ignoring multiple-day trips is that these may be multiple-site, multiple-length, and multiple-purpose trips, which makes it extremely difficult to estimate values for ecosystem services at specific sites. Ignoring multiple-day trips serves to underestimate the aggregate value that people who engage in recreation place on aquatic ecosystem services. Estimates for day trips can be affected by several key elements of any application. The first is the researcher's choice of the measurement of travel cost including the opportunity cost of travel time. A subjective decision by an analyst to include or exclude elements from the measurement of travel cost will increase or decrease the measurement of travel cost and affect value estimates.

The second factor is of particular concern for applications to aquatic ecosystems is that the degree to which aquatic ecosystem services are correlated with each other and with other physical attributes of a site. This multicollinearity makes it difficult to identify aquatic ecosystem attributes that people value and omitting relevant ecosystem attributes may lead to biased estimates. For example, if the environmental variable is binary and represents the presence of native trout and native trout occur in beautiful mountain streams, then the value estimate for native trout may also capture a value for scenic beauty. On the other hand, if a fish consumption advisory is place on an industrial river and is modeled as a binary variable in the RUM, then the value of removing the fish consumption advisory may also capture the value of fishing at a nonindustrial location.

A third key element affecting the quality of an application is the lack of consistent data on attributes that measure the same given attribute across all the sites in the choice set. Most of the RUMs employ the small set of attributes that are available for all sites. A related issue is the distinction between objective and subjective measures of site attributes—what matters is not how the attributes are measured by the experts but how they are perceived by the individual making the choice of recreation sites. It is much harder to obtain data on perceptions of site attributes.

### *Averting Behavior Models*

Averting behavior models have been increasingly used as an indirect method to evaluate the willingness of individuals to pay for improved health or to avoid undesirable health consequences (Dickie, 2003). In terms of aquatic ecosystems there are only two notable averting behavior applications: (1) a study of averting behavior in the presence of a waterborne disease giardiasis (Harrington et al., 1989) and (2) groundwater contamination by the solvent tricholoroethylene (TCE; Abdalla et al., 1992).

Averting behavior models are based on the presumption that people will change their behavior and invest money to avoid an undesirable health outcome. Thus, averting behavior analyzes the rate of substitution between changes in behavior and expenditures on and changes in environmental quality in order to infer the value of certain nonmarketed environmental attributes (see Appendix B). For example, in the presence of water pollution, a household may install a filter on the primary tap in the house to reduce or remove the pollutant. This involves a capital expenditure by the household and changes in behavior because potable water can now be safely obtained only from the primary tap, not from other taps in the house. Rather than producing a fishing trip or other type of recreational experience, as is the household production that underlies the estimation of a RUM, the household production here is protection from an undesirable outcome that is commonly health-related (Bartik, 1988; Courant and Porter, 1981; Cropper, 1981).

The giardiasis study by Harrington et al. (1989) is one of the best known avertingbehavior applications and one of the few applied to water. This study differs conceptually from the replacement cost studies for public water supplies discussed in Chapter 5, which are not based on individual preferences. The approach here is to measure people's actual averting expenditures to estimate a household value for avoiding an undesirable situation (i.e., contaminated drinking water). The model was applied to estimate the losses due to an outbreak of waterborne giardiasis in Luzerne County, Pennsylvania, that took place from 1983 to 1984. The outbreak occurred as a result of microbial contamination of the reservoir supplying drinking water to households in Luzerne County. Such contamination is typically caused by the ingestion of cysts of the enteric protozoan parasite *Giardia lamblia*, which is often found in animal (and sometimes human) feces deposited in upland watersheds that are subsequently transported to reservoirs used a source of drinking water. During the nine-month period of the Luzerne County outbreak, households were advised to boil their drinking water, but many also bought bottled water at supermarkets or collected free water supplied by some public facilities. The authors' "best estimate" of the average costs of these actions taken to avoid contaminated water ranged from \$485 to \$1,540 per household, or \$1.13 to \$3.59 per person per day for the duration of the outbreak.

In another averting behavior study conducted in Pennsylvania, Abdalla et al. (1992) investigated behavior by the Borough of Perkaise due to TCE in well water. Of the households in the borough, 43 percent indicated that they were aware of TCE in their water and 44 percent undertook actions to avoid exposure. The averting actions included purchasing bottled water, installing a home water treatment system, obtaining water from an uncontaminated source, and boiling water. Each of these actions required households to change their behavior and make out-of-pocket expenditures. The investigators found that households were more likely to undertake averting behavior if their perceived risk of consuming water with TCE was higher, if they knew more about TCE, or they had children the household had between the ages of 3 and 17. Of the households that averted, those with children less than three years of age spent more on averting activities than did other households. The average daily expenditure per household undertaking averting behaviors was about $0.06 during the 88 weeks that the TCE contamination persisted.

For an averting behavior study on water quality to be successful, four conditions are necessary:

1. households must be aware of compromised water quality;
2. households must believe that the compromised water quality will adversely affect the health of at least one household member;
3. there must be activities that a household can undertake to avoid, or reduce exposure to, the compromised water; and
4. households must be able to make expenditures that result in optimal protection.

The fourth element is rarely met however, so that total expenditures generally underestimate value and marginal expenditures should cautiously be interpreted as a measure of marginal willingness to pay.

Thus, an averting behavior study provides an estimate of the value households place on improving water quality. However, averting behavior studies rarely provide estimates of economic values of ecosystem services as defined in Chapter 2 and the beginning of this chapter. Averting expenditures generally are not the same as subtractions to income that leave people equally satisfied from an economic perspective as they would be if water quality were not improved. Averting behavior can underestimate or overestimate this value. An averting-behavior study would underestimate the economic value of clean water because averting behavior studies do not include the inconvenience of having to undertake the averting behavior. Economic value can also be underestimated if households cannot fully remove the diminished water quality. For example, onsite reverse osmosis treatment systems do not fully mitigate arsenic in drinking water (EPA, 2000b; Sargent-Michaud and Boyle, 2002). Averting behavior overestimates economic values when joint production is present, which could arise when contamination is present and the natural taste of the water is undesirable. Averting behavior would be undertaken to avoid the contamination and to obtain potable (more palatable) water. In this case, averting expenditures overstate what would be spent just to avoid the contamination.

Although averting behavior studies will generally provide a lower or upper bound on the damages to compromised drinking water, they are not likely to be useful in measuring other economic values of aquatic ecosystem services. Certainly, potable water is an important service of aquatic ecosystems to humans. Protected water for human consumption will have additional benefits of the clean water for other living organisms. As with RUMs, modeling is needed to understand how actions taken to protect or improve aquatic ecosystems will affect potable water.

### *Hedonic Methods*

Hedonic methods analyze how the different characteristics of a marketed good, including environmental quality, might affect the price people pay for the good or factor. This type of analysis provides estimates of the implicit prices paid for each characteristic. The most common application of hedonic methods in environmental economics is to real estate sales (Palmquist, 1991, 2003; Taylor, 2003). For example, the hedonic price function for residential property sales might decompose sale prices into implicit prices for the characteristics of the lot (e.g., acreage), characteristics of the house (e.g., structural attributes such as square footage of living area), and neighborhood and environmental quality characteristics. In terms of aquatic ecosystems, properties with lake frontage sell for more than similar properties that do not have lake frontage. Among properties with lake frontage, those located on lakes with good water quality would be expected to sell for more than those located on lakes with poor water quality. In thisz regard, a hedonic analysis is simply a statistical procedure for disentangling estimates of the premium people pay for lake frontage or for higher water quality, which is the revealed value for these ecological services.

There are two stages in the estimation of a hedonic model (Bartik, 1987; Epple, 1987). The first stage, which is commonly undertaken, simply decomposes sale prices of properties to estimate the implicit prices of property characteristics as described above. The implicit price estimates provide the marginal prices that people would pay for a small change in each characteristic. For example, if the attribute of interest was feet of frontage that the property had on a lake, the first-stage analysis provides the implicit price of a 1-foot increase in frontage. What if the policy question was how much value 100 feet of frontage would add to a property? However, the marginal price cannot provide this value estimate. The second-stage analysis uses either restrictions on the underlying utility function to derive value estimates (Chattopadhayay, 1999) or implicit price estimates from a number of different lakefront markets (Palmquist, 1984).

The application of a hedonic analysis requires a large number of property sales where characteristics of the properties vary. For example, data from a single lake might be used to estimate a first-stage equation for lake frontage if the amount of frontage varies for different properties on the lake. However, data from one lake probably cannot be used to estimate the value of water quality because all properties on a lake likely experience the same level of water quality. To estimate an implicit price for water quality it is necessary to have sales from a number of different lakes that differ in ambient water quality.

In order to operationalize a hedonic model to estimate values for aquatic ecosystem services, it must be assumed that buyers and sellers of properties have knowledge of the services and have access to the same information. For example, one problem in examining the effects of water pollution on property prices is that the use of water quality indices developed by natural scientists to measure pollutants, such as dissolved oxygen, nitrogen, and phosphorus, may not provide relevant information. As such, the physical measures of quality are not observable to homeowners, test results may not be generally available or easily obtained, and diminished water quality may not directly impair the enjoyment that households derive from waterfront homes (Leggett and Bockstael, 2000). Consider groundwater contamination as an example. The water that comes through a household tap may appear clean and taste fine but, if contaminated (e.g., by arsenic), may not be safe to drink. A hedonic model can be operational only if buyers and sellers are aware of arsenic levels in tap water and what levels are considered safe. Such information would be available if the public were generally aware of arsenic contamination, if sellers were required to reveal test results, or if buyers were advised to have the water tested if test results were not provided by the seller. In this example, since there is no obvious clue to the public that water quality is compromised, public information is necessary to prompt buyers and sellers to react to potential contamination. Another example is eutrophication of lakes. Although buyers and sellers cannot directly observe elements of the water chemistry that is compromised, they can certainly observe the physical manifestations of eutrophication. Thus, a summary measure of eutrophication (e.g., Secchi disk measurement of water clarity; see more below) may more be more closely aligned with buyer and seller perceptions than actual measures of water chemistry. This means that Secchi disk measurements may do a better job of explaining changes in sale prices of properties than measurements of dissolved oxygen, which implies a more accurate estimate of the implicit price placed on eutrophication by homeowners.

As noted above, most hedonic studies just estimate the first-stage, hedonic price function. Several of these studies have estimated implicit prices for water and coastal quality in the Chesapeake Bay area (Feitelson, 1992; Leggett and Bockstael, 2000; Parsons, 1992). Leggett and Bockstael (2000) showed that the concentration of fecal coliforms (a commonly used bacterial indicator of the potential presence of waterborne pathogens; see also NRC, 2004) in water has a significant effect on property values along the bay. They found that a change in fecal coliform counts of 100 colony forming units (CFUs) of water per 100 mL would affect sale prices of properties by about 1.5 percent, with the dollar amount ranging from about $5,000 to nearly $10,000. The average sale prices of properties in the study were $378,000 dollars, and the fecal contamination index ranged from 10 to 1,762, with a mean of 108 CFUs.

Parsons (1992) used a repeated-sale analysis to observe price changes on houses sold before and after the State of Maryland imposed building restrictions in critical coastal areas of the Chesapeake Bay. Prices for waterfront properties increased by 46-62 percent due to the restrictions, between 13 and 27 percent for houses nearby but not on the waterfront, and between 4 and 11 percent for houses as far as 3 miles away. Parsons noted however, that the price increases may be due to the increasing scarcity of near-coastal land as a result of the state restrictions. The Parsons study is interesting for two reasons. First, although a water quality attribute does not directly enter the hedonic price function, the benefits of the building restrictions include protection of aquatic and related coastal ecosystems along the coast. However, the second interesting feature is a complication of many hedonic studies—that environmental attributes may be highly correlated. Thus, it may be impossible to statistically disentangle the implicit price for the protection of aquatic ecosystems along the coast and other benefits of building restrictions.

Other applications of hedonic models to estimate implicit prices for aquatic ecosystems include the following:

- effects of water clarity on sale prices of lakefront properties (Michael et al., 2000; Steinnes, 1992; Wilson and Carpenter, 1999);
- effect of the potential for surface water contamination on farmer purchases of herbicides (Beach and Carlson, 1993);
- proximity of properties to hazardous waste sites that pollute groundwater (Kiel, 1995);
- extent of aquatic area proximate to properties (Paterson and Boyle, 2002);
- proximity of properties to wetlands (Doss and Taff, 1996; Mahan et al. 2000);
- effects of various measures of lake water quality (e.g., summer turbidity, chlorophyll concentrations, suspended solids, dissolved oxygen) on sale prices (Brasheres, 1985);
- effect of minimum lake frontage on sale prices of property to preserve lake amenities (Spalatro and Provencher, 2001);
- effect of coastal beach pollution on property prices (Wilman, 1984); and
- effect of pH levels in streams on property sale prices (Epp and Al-Ani, 1979).

A notable consideration of these studies is that the services of aquatic ecosystems have been included in the first-stage hedonic price equations in three ways. The first is a measure of the ecosystem quality as it affects the desirability of human use. The second is simply proximity to the aquatic ecosystem, and the third, which has been made possible with enhanced geographic information system (GIS) databases, measures the physical size of an aquatic ecosystem. All of the listed studies assessed surface water, with a primary focus on water quality in lakes. Furthermore, the Beach and Carlson (1993) study was the only hedonic analysis that considered an aquatic ecosystem that was not based on sales of residential properties.

Only one study has estimated the second-stage demand for an aquatic ecosystem service. Boyle et al. (1999) estimated the demand for water clarity in lakes using the multiple-market method. Clarity is measured by the depth at which a Secchi disk[6] disappears from sight as it is lowered into the water. Given an initial clarity reading of 3.78 meters, an increase in clarity to 5.15 meters results in a one-time value estimate of about $4,000 per household. Conversely, a decline of clarity from 3.78 meters to 2.41 meters results in a loss of value of at least $25,000 per household.

While hedonic models provide a useful method of estimating values for aquatic ecosystem services, the collinearity of attributes in hedonic price equation is a serious issue. In the Michael et al. (2000) study, Secchi disk measurements were used as a summary measure of lake eutrophication that is observable to property owners. Other lake attributes are highly correlated with reduced Secchi disk measurements, such as lake area and lake depths and small shallow lakes are more likely than larger lakes to be eutrophic. Eutrophic lakes are also typically warmer than oligotrophic lakes for swimming and support warm-water species of sportfish, including bass and perch that are typically less desirable than trout and salmon. Thus, although the Secchi disk measurements are a summary measure of water quality, it is likely that estimated implicit prices include the effects of other lake attributes on sale prices.

For a hedonic study to be operational there are two important conditions: (1) the effects of aquatic ecosystems must be observable to property owners, and (2) there should be minimal correlation between aquatic ecosystem services that affect sale price of properties and other attributes that affect sale prices.

A key feature in the modeling of aquatic ecosystem services is that the variable included in the hedonic price equation to reflect the ecosystem service being valued must be observable to property owners. As noted above, measured elements of water chemistry such as dissolved oxygen and chlorophyll levels may be less important than a summary measure such as Secchi disk readings. However, there still remains a question of whether homeowners' subjective perceptions of clarity are a better measure of service quality than physical Secchi disk measures. Poor et al. (2001) demonstrated that Secchi disk measurements of water clarity do a better job of explaining differences in sale prices than did property owners subjective ratings of water clarity. Thus, while aquatic ecosystem characteristics must be observable to homeowners, some type of objective measure of the characteristics is likely to be better than self-reports of the quantity or quality of services by homeowners. Finally, as long as aquatic ecosystem services are correlated with other attributes of property, hedonic analyses are likely to overestimate implicit prices and values.

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[6] A Secchi disk is most commonly an 8-inch metal disk painted with alternating black and white quadrants and is used to see how far a person can into the water (see *http://www.mlswa.org/secchi.htm* for further information).

## Production Function Methods

Production function (PF) approaches, also called "valuing the environment as input," assume that an environmental good or service essentially serves as a factor input into the production of a marketed good that yields utility. Thus, changes in the availability of the environmental good or service can affect the costs and supply of the marketed good, the returns to other factor inputs, or both. Applying PF approaches therefore requires modeling the behavior of producers and their response to changes in environmental quality that influence production (see Appendix C for further information about the general PF approach). Dose-response and change-in-productivity models, which have been used for some time, can be considered special cases of the PF approach in which the production responses to environmental quality changes are greatly simplified.

However, more sophisticated PF approaches are being increasingly employed for a diverse range of environmental quality impacts and ecosystem services, including the effects of flood control, habitat-fishery linkages, storm protection functions, pollution mitigation, and water purification. A two-step procedure is generally invoked (Barbier, 1994). First, the physical effects of changes in a biological resource or ecological service on an economic activity are determined. Second, the impact of these environmental changes is valued in terms of the corresponding change in the marketed output of the relevant activity. In other words, the biological resource or ecological service is treated as an "input" into the economic activity, and like any other input, its value can be equated with its impact on the productivity of any marketed output.

For some ecological services that are difficult to measure, an estimate of ecosystem area may be included in the production function of marketed output as a proxy for the ecological service input. For example, in models of coastal habitat-fishery linkages, allowing wetland area to be a determinant of fish catch is thought to "capture" some element of the economic contribution of this important ecological support function (Barbier and Strand, 1998; Barbier et al., 2002; Ellis and Fisher, 1987; Freeman, 1991; Lynne et al., 1981). That is, if the impacts of the change in the wetland area input can be estimated, it may be possible to indicate how these impacts influence the marginal costs of production. As shown in Figure 4-1, for example, an increase in wetland area increases the abundance of crabs and thus lowers the cost of catch. The value of the wetlands support for the fishery—which in this case is equivalent to the value of increments to wetland area—can then be imputed from the resulting changes in consumer and producer value.

For the PF approach to be applied effectively, it is important that the underlying ecological and economic relationships are well understood. When production is measurable and either there is a market price for this output or one can be imputed, determining the marginal value of the ecological service is relatively straightforward. If the output of the affected economic activity cannot be measured directly, then either a marketed substitute has to be found or possible complementarity or substitutability between the ecological service and one or more of the other (marketed) inputs has to be explicitly specified. All of these applications require detailed knowledge of the physical effects on production of changes in the ecological service. However, applications that assume complementarity or substitutability between the service and other inputs are particularly stringent in terms of the information required on physical relationships in production. Clearly, cooperation is required between economists, ecologists, and other researchers to determine the precise nature of these relationships.

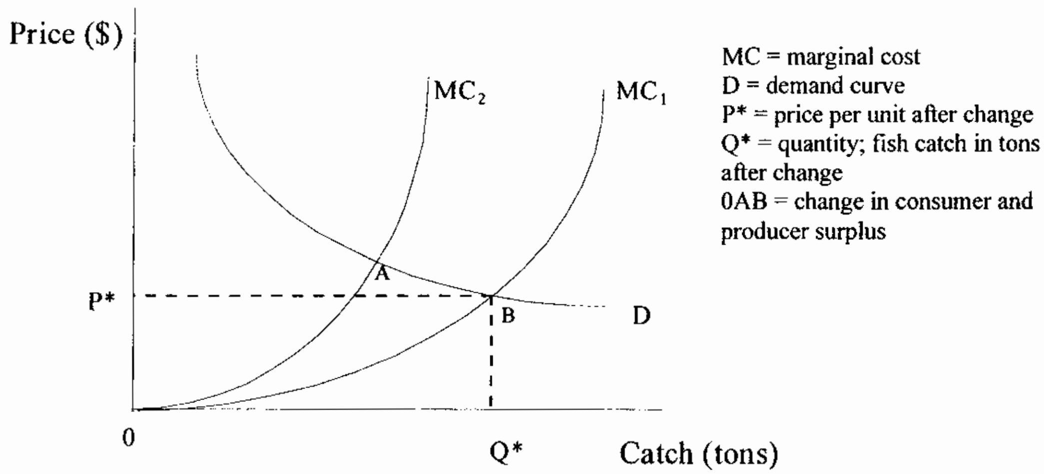


FIGURE 4-1 The economic value effects of increased wetland area on an optimally managed fishery. For optimally managed fishery a change in wetland area that serves as a breeding ground and nursery results in a shift in the marginal cost curve (MC) of the fishery. The welfare impact is the change in consumer and producer surplus (represented by area 0AB). SOURCE: Adapted from Freeman (1991).

In addition, as pointed out by Freeman (1991), market conditions and regulatory policies for the marketed output will influence the values imputed to the environmental input. For instance, in the previous example of coastal wetlands supporting an offshore crab fishery, the fishery may be subject to open-access conditions. Under these conditions, profits in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer values are zero and only consumer values determine the value of increased wetland area (see Figure 4-2).

A further issue is whether a static or dynamic model of the relationship between the ecological service and the economic activity is required. As discussed in Appendix B, this usually depends on whether or not it is more appropriate to characterize this relationship as affecting production of the economic activity over time. Figures 4-1 and 4-2 represent PF models that are essentially static. The value of changes in the environmental input is determined through producer and consumer value measures of any corresponding changes in the one-period market equilibrium for the output of crabs. In dynamic approaches, the ecological service is considered to affect an intertemporal, or "bioeconomic," production relationship. For example, a coastal wetland that serves as a breeding and nursery habitat for fisheries could be modeled as part of the growth function of the fish stock, and any value impacts of a change in this habitat support function can be determined in terms of changes in the long-run equilibrium conditions of the fishery or in the harvesting path to this equilibrium (see Appendix B). Figure 4-3 shows that the long-run supply curve for an open-access fishery is typically backward-bending (Clark, 1976). Since coastal wetland habitat affects the biological growth of the fishery, a decline in wetland area will shift back the long-run supply curve of the fishery and thus reduce long-run harvest levels. The corresponding losses can be measured by the fall in economic value, which will be greater if the demand curve is more inelastic (i.e., steeper).

A number of recent studies have used PF models to estimate the economic benefits of coastal wetland-fishery linkages. Much of this literature owes its development to the approach

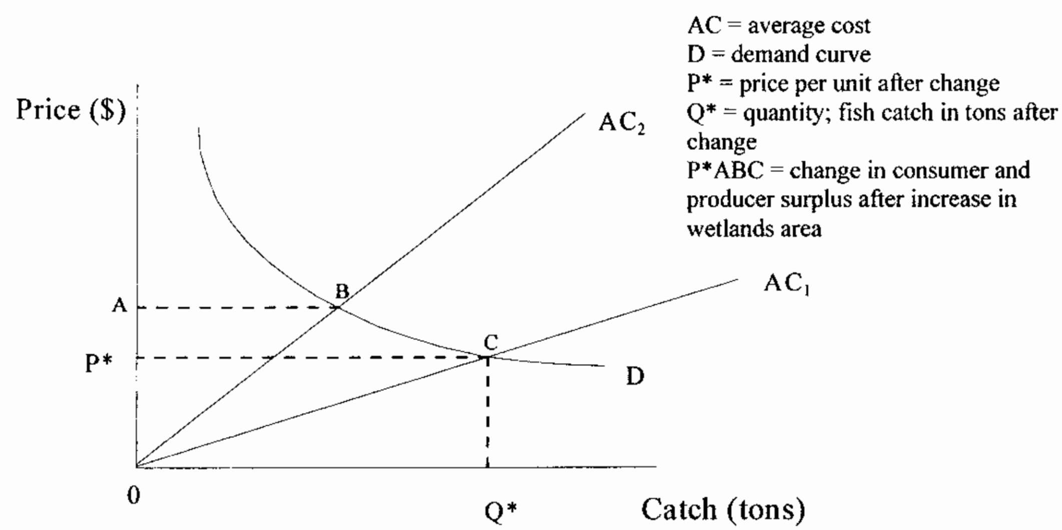


FIGURE 4-2 The economic value effects of increased wetland area on an open-access fishery. For open-access fishery, a change in wetland area that serves as a breeding ground and nursery results in a shift in the average cost curve, AC, of the fishery. The welfare impact is the change in consumer surplus (area P*ABC). SOURCE: Adapted from Freeman (1991).

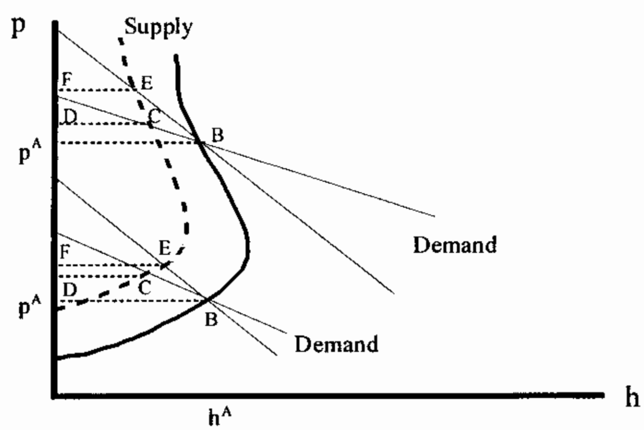


FIGURE 4-3 Wetland loss and the long-run market equilibrium of an open-access fishery.
The effect of a fall in wetland area is to shift the long-run equilibrium supply curve of an open access fishery to the left. The result is a decline in fish harvest $h^A$. The loss in consumer value will be greater if the demand curve is more inelastic (area $P^ABEF$) than elastic (area $P^ABCD$). SOURCE: Adapted from Barbier et al. (2002).

of Lynne et al. (1981) who suggested that the support provided by the marshlands of southern Florida for the Gulf Coast fisheries could be modeled by assuming that marshland area supports biological growth of the fishery. For the blue crab fishery in western Florida salt marshes, the authors estimated that each acre of marshland increased productivity of the fishery by 2.3 pounds per year. Others have applied the Lynne et al. approach to additional Gulf Coast fisheries in western Florida (Bell, 1997) and in southern Louisiana (Farber and Costanza, 1987). Using data from the Lynne et al. (1981) case study, Ellis and Fisher (1987) determined the impacts of changes in the Florida Gulf Coast marshlands on the supply-and-demand relationships of the commercial blue crab fishery. They demonstrated that an increase in wetland area increases the abundance of crabs and thus lowers the cost of catch. The value of the wetlands' support for the fishery—which in this case is equivalent to the value of increments to wetland area—can then be imputed. Freeman (1991) has extended Ellis and Fisher's approach to show how the values imputed to wetlands are influenced by market conditions and regulatory policies that affect harvesting decisions in the fishery. In assuming an open-access crab fishery supported by Louisiana coastal wetland habitat, the value of an increase in wetland acreage from 25,000 to 100,000 acres could range from $47,898 to $269,436. If the fishery is optimally managed, the increase in coastal wetland is valued from $116,464 to $248,009.

More "dynamic," or long-term, approaches to analyzing habitat-fishery linkages have also been developed (e.g., see Barbier and Strand, 1998; Barbier et al., 2002; Kahn and Kemp, 1985; McConnell and Strand, 1989). For example, in their case study of valuing mangrove-shrimp fishery linkages in the coastal regions of Campeche, Mexico, Barbier and Strand (1998) analyzed the effects of a change in mangrove area in terms of influencing the long-term equilibrium of an open-access fishery (i.e., one in which there are no restrictions on additional fishermen entering to harvest the resource). Their results indicate that the economic losses associated with mangrove deforestation appear to vary with long-term management of the open-access fishery. During the first two years of the simulation (1980-1981), which were characterized by much lower levels of fishing effort and higher harvests, a 1 $km^2$ decline in mangrove area was estimated to reduce annual shrimp harvests by around 18.6 tons, or a loss of about $153,300 per year. In contrast, during the last two years of the analysis (e.g., 1989-1990), which saw much higher levels of effort and lower harvests in the fishery, a marginal decline in mangrove area resulted in annual harvest losses of 8.4 tons, or $86,345 each year.

Kahn and Kemp (1985) and McConnell and Strand (1989) considered the impacts of water quality on fisheries in the Chesapeake Bay. Kahn and Kemp related the environmental carrying capacity of fish populations to the level of subaquatic vegetation, which is in turn affected by the runoff of agricultural chemicals, discharges from waste treatment plants, and soil erosion. Based on this analysis, the authors were able to determine marginal and total damage functions for various finfish and shellfish species in the bay.

Swallow (1994) modeled the impacts of developing "high-quality" and "normal-quality" freshwater pocosin (peat-bog) estuarine wetlands on the Pamlico Sound, North Carolina, shrimp fishery. Drainage of the pocosin wetlands for forestry and agricultural uses irreversibly alters the local hydrological system by eliminating the vegetative and peat-bog structure that inhibits water flow, causing a decline in the salinity of the estuarine shrimp nursery areas. The result is a decline in the juvenile shrimp stock necessary to replenish the Pamlico Sound fishery each year. Through his production function model linking development to salinity changes in the pocosin and fishery declines, Swallow estimated that the greatest losses to the shrimp fishery are estimated as $3.37 per acre per year for developing agriculture that affects "high-quality" wetlands near the southwestern shore of the sound. However, losses in other areas of the estuary with normal-quality wetlands are much lower. Based on these estimates, Swallow was able to determine the net opportunity cost of development of different-quality wetlands in the sound. The efficient policy would be to halt agricultural development when the marginal value of development net of the offshore fishery impacts fell to an annualized $1.12 per acre ($14 in present value). For the pocosin wetlands of the sound, this implies that 9,800 of the 11,009 acres of normal-quality southeastern wetlands could be safely developed, but all 1,209 high-quality southwestern wetlands should be preserved.

As these preceding examples illustrate, most uses of the production function approach have been concerned with valuing single ecosystem services. However, there have been a number of recent attempts to extend this approach to the ecosystem level through integrated economic-ecological modeling. The PF approach has the advantage of capturing more fully the ecosystem functioning and dynamics underlying the provision of key services and can be used to value multiple services arising from aquatic ecosystems.

For example, Wu et al. (2003) examined the effectiveness of alternative salmon habitat restoration strategies in the John Day River Basin, Oregon, through employment of integrated biological, hydrologic, and economic models. The purpose of the modeling was to shed light on two sets of unknown factors affecting salmon restoration investments: (1) the effects of uncertain environmental factors, such as weather and ocean conditions; and (2) the limited information on the potential ecological and hydrological threshold effects that can affect the potential payoffs on restoration investments. In an ideal salmon habitat, stream temperature must be below a certain threshold level. When water temperature exceeds this level, reducing temperature by one or two degrees will have no impact on fish survival. Other ecological factors, such as streamside vegetation, soil sedimentation, and species interaction, should also be modeled to examine trade-offs between different conservation benefits through investments targeted at one benefit (e.g., salmon habitat restoration). For example, Wu and colleagues demonstrated that for cold water-adapted fish species (e.g., rainbow trout, Chinook salmon), provided water temperature is maintained below its critical threshold, the number of fish increases as the vegetative use index improves. However, for speckled dace, the number of fish per kilometer of stream decreases as vegetative use improves and temperature decreases. In their fully integrated model, the authors were able to show the trade-offs of different salmon restoration investments in terms of the decline of speckled dace and the estimated marginal social value of increased numbers of cold water fish species. This is a trade off between quantity in one aspect of the ecosystem and quality in another aspect. A three-degree drop in stream temperature, from 26°C to 23°C, will result in an estimated social benefit of $22,129 from increases in cold-water sportfish species, but a reduction of 506 speckled dace per kilometer of stream.

Carpenter et al. (1999) demonstrated how an integrated ecological-economic model of eutrophication of small shallow lakes can demonstrate the value impacts of irreversible ecological change (see also Chapter 5). Tschirhart and Finhoff (2001) developed a general equilibrium ecosystem with a regulated open-access fishery to analyze simulations of an eight-species Alaskan marine ecosystem that is affected by fish harvesting. Fishing impacts the commercial fish population as well as the populations of other species, including Steller sea lions, an endangered species. Settle and Shogren (2002) developed an integrated ecological-economic model to analyze the impacts of the introduction of exotic lake trout into Yellowstone Lake, which pose a risk to the native cutthroat trout. The authors demonstrated that an integrated model leads to different policy results than treating the ecological and economic systems separately. Under the best case scenario, the U.S. Park Service eliminates lake trout immediately and without cost, while under the worst-case scenario lake trout are left alone. An integrated model has little effect on the worst-case scenario, because the likely outcome is elimination of cutthroat trout. However, under the best-case scenario without feedbacks, the steady-state population of cutthroat trout is about 2.7 million. With feedbacks, the steady-state population is about 3.4 million. The integrated model predicts that the maximum optimal fixed budget for lake trout control is \$169,000.

Other applications of production function models to estimate the value of services of aquatic ecosystems include the following:

- habitat-fishery linkages (Barbier, 2000 and 2003; Batie and Wilson, 1978; Bell, 1989; Costanza et al., 1989; Danielson and Leitch, 1986; Hammack and Brown, 1974; Sathirathai and Barbier, 2001);
- coastal erosion control and storm protection (Costanza and Farber, 1987; Costanza et al. 1989; Sathirathai and Barbier, 2001);
- groundwater recharge of wetlands (Acharya 2000; Acharya and Barbier 2000; 2002);
- water quality-fishery linkages (Kahn, 1987; Loomis, 1988; Wu et al., 2000); and
- general equilibrium modeling of integrated ecological-economic systems (Tschirhart, 2000).

## Stated-Preference Methods

Stated-preference methods have been commonly used to value aquatic ecosystem services. There are two variants of stated-preference methods, contingent valuation (e.g., Bateman et al., 2002; Boyle, 2003; Mitchell and Carson, 1989) and conjoint analysis (e.g., Holmes and Adamowicz, 2003; Louviere, 1988; Louviere et al., 2000). Contingent valuation was developed by economists and is the more commonly used approach, whereas conjoint analysis was developed in the marketing literature (Green and Srinivasan, 1978). Contingent valuation attempts to measure the value people place on a particular environmental item taken as a specific bundle of attributes; conjoint analysis aims to develop valuation functions for the component attributes viewed both separately and in alternative potential combinations.

Contingent valuation is used to estimate values for applications, such as aquatic ecosystem services, where neither explicit nor implicit market prices exist. The first known application of contingent valuation was by Davis (1964) for hunters and other visitors to the Maine woods. About 10 years later, the third application of contingent valuation (Hammack and Brown, 1974) estimated the value of waterfowl and wetlands. Through the 1980s and 1990s, the quality and extent of contingent valuation studies appear to have increased steadily.

While conjoint analysis was developed in the marketing literature to estimate prices for new products or modifications of existing products, it is conceptually similar to contingent valuation, and economists have come to recognize that it is another stated-preference approach to estimating economic value when market prices are unavailable. The first known environmental application was by Rae (1983) to value air quality in national parks. The number of environmental applications of conjoint analysis increased throughout the 1990s.

Both contingent valuation and conjoint analysis use survey questions to elicit statements of value from people with two key distinctions. First, contingent valuation studies generally pose written or verbal descriptions of the environmental change to be valued, while conjoint analysis poses the change in terms of changes in the attributes of the item to be valued. Consider a wetland restoration project as an example—the Macquarie Marshes in New South Wales, Australia (Morrison et al., 1999; also discussed below). A contingent valuation survey would contain a description of the wetland in its current condition and the wetland after restoration, whereas a conjoint survey would describe the wetland in terms of key attributes. These might be acres of wetland, number of species of breeding birds, and frequency with which birds breed. A contingent valuation study may contain this same information, but it would not be presented to estimate component values for each of these attributes. In terms of valuation, the contingent valuation study provides an estimate of the value of change in the marsh due to restoration, while the conjoint study provides a similar estimate and also estimates the amount of value contributed by each attribute. Thus, like a hedonic model, the attribute-based approach of conjoint analysis provides implicit prices for key attributes of the aquatic ecosystem.

The second key difference between these stated-preference methods involves the response formats. Contingent valuation studies typically ask respondents to state their value directly or to indicate a range in which the value resides (Welsh and Poe, 1998). In the latter case, econometric procedures are used to estimate the latent value based on the monetary intervals that respondents indicate. In conjoint analysis, survey respondents would be give alternatives to consider (e.g., three marsh restoration programs) and asked to choose the preferred alternative or to rank the alternatives (Boyle et al., 2001). Again, econometric procedures are used to estimate values from the choices or ranks.

Of the many contingent valuation studies that have been conducted, perhaps the two most well known involve aquatic ecosystems. In one of the earliest large-scale, contingent valuation studies, Mitchell and Carson (1981) estimated total national values for inland waters that are swimmable, fishable, and drinkable. They found that people who use freshwater for recreation were willing to pay $237 annually to obtain swimmable, fishable, and drinkable freshwater, while the comparable estimate for nonusers was $111.

The second study examined the value that a national sample would place on protecting Prince William Sound from an oil spill of the magnitude of the *Exxon Valdez* spill (Carson et al., 1992). In this study, a national survey was also conducted and total values were estimated, although the estimates were assumed to be primarily nonuse values because most people in the nationwide sample would never actually visit Price William Sound. The median value estimated was about $33 per household for a one-time payment to protect Prince William Sound from a large-scale oil spill.

Many contingent valuation studies have investigated values for aquatic ecosystem services. So many, in fact, that several meta-analyses of these studies have been conducted, including protection of groundwater from contamination (Boyle et al., 1994); wetland values (Woodward and Wui, 2001); and sportfishing (Boyle et al.,1998 a,b).

The primary application of the contingent valuation groundwater studies is protection from nitrate contamination resulting from agricultural practices. A particularly interesting attribute of the wetland meta-analysis is that the authors attempted to determine how values for wetlands vary with the services they provide. Lastly, the vast majority of sportfishing contingent valuation studies have investigated values of a single-day fishing trip—some focusing on individual species and others addressing some type of contamination.

The use of conjoint analysis is relatively new for nonmarket valuation and very few conjoint studies of aquatic ecosystems services have been undertaken. The best example is the aforementioned study of the Macquarie Marshes by Morrison et al. (1999). This study found that households in the area of New South Wales, Australia (near the marshes) would pay about $150 (Australian dollars) per year to restore the marshes to part of their original area. This change included increasing the number of species of marsh birds and the frequency at which they breed (Morrison and Boyle, 2001). Other examples include waterfowl hunting (Gan and Luzar, 1993), and salmon fishing (Roe et al., 1996). The use of conjoint analysis in other types of applications in the literature is growing, and conjoint analysis is likely to become more prominent in the valuation of aquatic ecosystems in the future because of its ability to estimate values for multiple services. Most aquatic ecosystems provide multiple services (see also Chapter 3), and the ability to estimate marginal values for specific services is important for policy analyses.

To implement a stated-preference study two key conditions are necessary: (1) the information must be available to describe the change in an aquatic ecosystem in terms of service that people care about, in order to place a value on those services; and (2) the change in the aquatic ecosystem must be explained in the survey instrument in such a way that people will understand and not reject the valuation scenario. However, achieving these two conditions is easier said than done. Identifying the services that people care about with respect to a resource is not always a simple task because aquatic ecosystems such as wetlands provide a wide variety of services. People may care about wetland birds and animals and have no difficulty linking these to wetlands; however, potential respondents may have greater difficulty linking a wetland policy to changes in flood risk or the cost of potable water. Even if respondents identify and consider all relevant services, they may misinterpret policy descriptions or misperceive the impact of policy described in a questionnaire (Johnston et al., 1995; Lupi et al., 2002).

It is now common for valuation research to use qualitative methods to identify valued services and develop stated-choice questionnaires. Valuation questionnaires pose a cognitive problem to respondents, and the design of the questionnaire may facilitate or detract from respondents' solutions to the problem (Sudman et al., 1996; Tourangeau et al., 2000). Focus groups and individual interviews are both effective in understanding ecosystem services and the valuation problem from respondents' points-of-view (Johnston et al., 1995; Kaplowitz and Hoehn, 2001). Draft questionnaires may be tested and refined through individual pretest interviews, followed by careful debriefing by interviewers especially trained to identify questionnaire miscues (Kaplowitz et al., 2003).

The development of a questionnaire can be problematic with regard to obtaining the information necessary to explain the change in an aquatic ecosystem in lay terms. In the case of potential groundwater contamination, it may be difficult to develop the probability that an aquifer will become contaminated and even more difficult to inform individual survey respondents of the likelihood that their wells will become contaminated. Poe and Bishop (1999) demonstrated that this type of respondent-specific information is crucial to the development of valid value estimates. There are also cases in which respondents might reject a valuation scenario outright. Using Lake Onondaga in Syracuse, New York, as an example, the long-term contamination of this site and the severity of the contamination might lead survey respondents to reject any scenario that elicited values for cleaning up pollution damages.

Having noted and provided some examples of the limitations of stated-preference methods however, the vast number of stated-preference methods in the literature is testimony to the wide array of aquatic ecosystem applications in which contingent valuation and conjoint analysis can be employed. Nevertheless, it is also important to note that much of the criticism of stated-preference methods has arisen because they are not based on actual behavior (e.g., Diamond and Hausman, 1994; Hanemann, 1994; Portney, 1994). The debate has centered mainly on the validity of employing contingent valuation techniques to estimate nonuse values (NOAA, 1993). In contrast, the validity of conjoint estimates of value is a relatively unexplored area of research. However, there is a basic concern regarding the accuracy of stated-preference estimates of value. Do stated-preference methods result in overestimates of value? Studies conducted in controlled experimental settings suggest that both contingent valuation and conjoint methods may overestimate values (Boyle, 2003; Cummings and Taylor, 1998, 1999). Although this concern exists, the absolute magnitude of overestimation has not been established, nor has if been established that this error is any greater that the errors identified for stated-preference methods elsewhere in this chapter.

Another issue that has not received enough attention in the stated-preference literature concerns the accuracy of this approach and what level of accuracy is acceptable. Whereas stated-preference methods have been criticized because experimental design features affect value estimates, context effects have been largely ignored in revealed-preference studies. Some of the features that are problematic in stated-preference studies (e.g., information, sequencing, starting prices) also perturb markets (Randall and Hoehn, 1996). In fact, this is essentially the substance of the marketing literature. Thus, although stated-preference methods have been much maligned, revealed-preference methods have not received the comparable scrutiny that they should receive. This dichotomy of evaluation perspectives occurs simply because stated-preference methods are based on behavioral intentions, while revealed-preference methods are based on actual behavior.

The bottom line is that some real biases have been identified in contingent valuation studies, and many of these same biases carry over to conjoint studies. These biases imply that careful study design and interpretation of value estimates are required, but these biases do not appear to be specific to aquatic ecosystem applications.

### Pooling Revealed-Preference and Stated-Preference Data

A number of recent valuation studies have used both revealed-preference and stated-preference data to estimate values. These analyses have pooled travel-cost data with stated-preference data that asks respondents to reveal intended visitation under specific environmental conditions (Adamowicz et al., 1994; Cameron, 1992). Pooling involves taking data from different valuation methods and using the combined data, typically from two valuation methods, to estimate a single model of preferences. Travel-cost data provide information on people's actual choice to inform the model estimation, but respondents may not have experienced the new environmental condition to be valued. These studies have used a hypothetical scenario to elicit statements of behavior, not willingness to pay, if the new condition occurred. These stated behaviors are added to the travel-cost data to estimate the preference model. This type of stated-preference data is sometimes referred to as "behavioral intentions." Some studies have framed the behavioral intention questions similar to contingent valuation questions, and visitation—not a dollar value—is the requested response (Cameron, 1992). Other studies have framed the behavioral intention question in a conjoint framework, asking people to indicate what type of trip they would take given the levels of different trip attributes (Adamowicz et al., 1994). The advantage of data pooling is the consistency imposed by actual choices, and the stated-preference data allow for environmental conditions where revealed behavior does not exist.

Cameron et al. (1996) used data pooling to investigate the values people place on recreation in the rivers and reservoirs in the Columbia River Basin. Data pooling was necessary because the policy question required values for water levels that were not represented in the current management regime. They found that the average consumer value for a flow management that enhanced recreation was about $72 per person for the months of July and August. If, however, the management strategy changed to facilitating fish passage for migration and spawning, the consumer value estimate fell to $40.

Almost all of the data-pooling studies to date have been conducted in the context of valuing sportfishing on freshwater lakes and rivers. The primary motivation has been to develop values where long-term contamination precludes the use of revealed-preference data to estimate values for ecosystem losses or improvements. The committee feels that these types of valuation studies will become more prevalent in the future. The issues discussed for the travel-cost method and stated-preference methods still persist in these analyses. In addition, another important issue arises that can substantially affect value estimates. That is, the empirical investigator must decide what weight to place on the stated-preference data and the revealed-preference data in the model estimation. The existing literature has largely ignored this important issue.

## Benefit Transfers

It is impossible to discuss economic valuation methods without also discussing benefit transfers. A benefit transfer is the process of taking an existing value estimate and transferring it to a new application that is different from the original one (Boyle and Bergstrom, 1992). There are two types of benefit transfers, value transfers and function transfers. A value transfer takes a single point estimate, or an average of point estimates from multiple studies, to transfer to a new policy application. A function transfers uses an estimated equation to predict a customized value for a new policy application. Benefit transfers are commonly used in policy analyses because off-the-shelf value estimates are rarely a perfect fit for specific policy questions. The EPA, recognizing the practical need to conduct benefit transfer, has developed the only peer-reviewed guidelines for conduct of these analyses (EPA, 2000a).

However, the committee does not advocate the use of benefit transfers for many types of aquatic ecosystem service valuation applications. First, with the exception of a few types of applications (e.g., travel-cost and contingent valuation estimates of sportfishing values), there are not a lot of studies that have investigated values of aquatic ecosystem services. Second, most nonmarket valuation studies have been undertaken by economists in the abstract from specific information that links the resulting estimates of values to specific changes in aquatic ecosystem services and functions. Finally, studies that have investigated the validity of benefit transfers in valuing ecosystem services have demonstrated that this approach is not highly accurate (Desvouges et al. 1998; Kirchhoff et al., 1997; Vandenberg et al., 2001). Because benefit transfers involve reusing existing data, a benefit transfer does not provide an error bound for the value in the new application after the transfer. For these reasons, benefit transfer is generally considered a "second best" valuation method by economists. The three studies cited above not only investigate the accuracy of benefit transfer, but also provide an idea of how large the error might be in using a benefit transfer to value aquatic ecosystem services.

As stated previously, the purpose of this chapter is to lay out carefully the currently available basic valuation approaches, whereas the purpose of the report as a whole is to facilitate original research and studies that will develop a closer link between aquatic ecosystem functions, services, and value estimates that ultimately lead to improved environmental decision-making. The committee recommends that although benefit transfer is in common use, it should be employed with discretion and caution. Future research should focus on enhancing the reliability of off-the-shelf value estimates that are available for use in benefit transfer applied to valuing the services of aquatic ecosystems.

### Replacement Cost and Cost of Treatment

In circumstances where an ecological service is unique to a specific ecosystem and is difficult to value by any of the above methods, and there are no reliable existing value estimates elsewhere to apply the benefit transfer approach, analysts have sometimes resorted to using the cost of replacing the service or treating the damages arising from loss of the service as a valuation approach.

Such an approach to approximating the benefits of a service by the cost of providing it is not used exclusively in environmental valuation. For example, in the health economics literature this approach is referred to as "cost of illness" (Dickie, 2003). This involves adding up the costs of treating a patient for an illness as the measure of benefit. Such an approach is not preference-based and is not a measure of economic value. If the treatment is not fully successful, then the patient might be willing to pay even more to avoid or treat an illness. On the other hand, market disturbances, often caused by government policies, might create conditions where more service is provided than an individual is actually willing to pay for. This information should be on the cost side of the benefit-cost ledger, not counted as a benefit.

Because of the lack of data for many ecological services arising from aquatic ecosystems, valuation studies may consider resorting to a similar *replacement cost* or *cost of treatment* approach. For example, the presence of a wetland may reduce the cost of municipal water treatment for drinking water because the wetland system filters and removes pollutants. It is therefore tempting to use the cost of an alternative treatment method, such as the building and operation of an industrial water treatment plant, to represent the value of the wetland's natural water treatment service. As with the health example, this is not a preference-based approach, and does not measure value; it is the cost of providing the aquatic ecosystem service that people value.

In general, economists consider that the replacement cost approach to estimating the value of a service should be used with great caution if at all. However, Shabman and Batie (1978) suggest that this method can serve as a last resort "proxy" valuation estimation for an ecological service if the following conditions are met: (1) the alternative considered provides the same services; (2) the alternative used for cost comparison should be the least-cost alternative; and (3) there should be substantial evidence that the service would be demanded by society if it were provided by that least-cost alternative. In the absence of any information on benefits, when a decision has to be made to take some action, then treatment costs become a way of looking for a cost-effective policy action.

Chapter 5 (see also Chapter 6) provides a case study discussion of the provision of clean drinking water to New York City by the Catskills watershed, in which the decision to restore the watershed was based on a comparison of the cost of replacing the water purification services of the watershed with a new drinking water filtration system. Thus, this application of the replacement cost method appears to fulfill the criteria of appropriate use of this method for valuation as suggested by Shabman and Batie (1978).

## Summary of Valuation Approaches and Methods: Pros and Cons

Thus far, this chapter has discussed a variety of environmental valuation methods and provided some examples of their application to aquatic ecosystem services. Table 4-2 summarizes this discussion of nonmarket valuation method and approaches and their applicability to key aquatic ecosystem services. The last column in Table 4-2 is perhaps the most important link in moving from this chapter to Chapter 5 because it identifies ways that aquatic ecosystem services have been included in empirical valuation studies to date.

TABLE 4-2 Integrating Nonmarket Valuation Methods of Aquatic Ecosystem Applications

| Valuation Methods | Types of Values Estimated | Common Types of Applications | Ecosystem Services |
|---|---|---|---|
| Travel cost | Use | Recreational fishing | Site visitation<br>Fish catch rates<br>Fish consumption advisories |
| Averting behavior | Use | Human health | Waterborne disease<br>Toxic contamination |
| Hedonics | Use | Residential property | Proximity (distance) to aquatic ecosystems<br>Water clarity<br>Various measures of water chemistry (e.g., pH, dissolved oxygen)<br>Area of aquatic ecosystems proximate to a property |
| Production function | Use | Commercial and recreational fishing;<br>Hydrological functions;<br>Residential property;<br>Ecological-economic modeling of the effects of invasions | Habitat-fishery linkages<br>Water quality-fishery linkages<br>Habitat restoration<br>Groundwater recharge by wetlands<br>Biological invasions<br>Eutrophication<br>Storm protection |
| Stated preferences | Use and nonuse | Recreation,<br>Human health and any other activity, including passive use, that affects peoples' economic values | Groundwater protection<br>Wetland values<br>Sportfishing<br>Waterfowl hunting |
| Benefit transfer | Use and nonuse | Recreation and passive use | Sportfishing |

For revealed-preference methods, the key issue is whether ecosystem services affect peoples' behavior. If a service of an aquatic ecosystem does not affect peoples' choices, there are three alternative means of addressing this in a valuation analysis.

1. The service that does not affect site choice may affect a service that does affect site choice. In this case, ecological modeling is needed to establish the link between services, which is the essence of the production function approach.
2. Another valuation approach may be needed. For example, if a wetland provides filtration to yield potable groundwater, then a RUM is not the approach to capture this value. The value of potable groundwater might be better estimated using a hedonic model or a stated-preference study.
3. If currently available methods of economic valuation or ecological knowledge are not capable of modeling the ecosystem service relationship of interest, then consideration of the service has to be acknowledged outside the empirical benefit analysis.

Although the above conditions apply to all revealed-preference methods discussed in this chapter, they are best illustrated in conjunction with the production function approach. As discussed earlier, the production function approach is reliant on actual market behavior or value estimates from revealed-preference or stated-preference studies. This approach is important because many changes in important functions and service of aquatic ecosystems do not directly affect humans (e.g., water quality and habitat changes that influence coastal and riparian fisheries; eutrophication; biological invasions). The production function approach is therefore a means of identifying values for these indirect relationships. However, to date, the applicability of production function approaches has been limited to a few types of aquatic ecosystem services, such as habitat effects on fisheries, coastal erosion, lake habitat quality, and the resilience of aquatic systems to invasive species. There are two reasons for this. First, for this approach to be applied effectively, it is important that the underlying ecological and economic relationships are well understood. Unfortunately, our knowledge of the ecological functions underlying many key aquatic ecosystem services is not fully developed (see Chapter 3). Second, effective application of production function approaches also requires detailed knowledge of the physical effects on production of changes in the ecological service. Threshold effects and other nonlinearities in the underlying hydrology and ecology of aquatic systems, and the need to consider trade-offs between two or more environmental benefits generated by ecological services, complicate this task. Recent progress in developing dynamic production function approaches to modeling ecosystem services, such as habitat-fishery linkages and integrated ecological-economic analysis to incorporate multiple services and environmental benefit trade-offs, have illustrated that the production function approach may have a wider application to valuing the services of aquatic ecosystems as our knowledge of the ecological, hydrological, and economic features of these systems improves.

In comparison to revealed-preference methods, stated-preference methods exhibit the following advantages, they are: (1) the only methods available for estimating nonuse values; (2) employed when environmental conditions have not or cannot be experienced so that revealed-preference data are not available; and (3) used to estimate values for ecosystem services that do not affect peoples' behavior.

The first advantage is quite obvious, nonuse values by definition do not have a behavioral link that would allow a revealed-preference method to be employed. People do not have to exhibit any type of use behavior or monetary transaction to hold nonuse values. More importantly, a second advantage of stated-preference approaches is that they can be employed in situations where people may not have experienced the new environmental condition. For example, Lake Onondoga in New York has experienced sufficient long-term contamination to preclude uses such as fishing and swimming. Thus, it would be impossible to estimate travel-cost models for these activities. However, it might be possible to develop a stated-preference survey to elicit values if it were possible to improve water quality in the lake. Finally, there may be ecosystem services that serve important ecological functions (see Tables 3-2 and 3-3), but do not affect peoples' use of aquatic ecosystems in a directly observable manner. If the ecological link were explained to people it might be possible to use a stated-preference study to elicit values for such services. For example, people might not understand the role that wetlands play in the purification of groundwater recharge from surface waters. It would be possible, however, to design a stated-preference study to elicit values for the protection of wetlands to protect water purification services.

Despite these advantages of stated-preference methods, the above discussion highlights a number of concerns and problems identified in the literature, including issues of identifying the relevant ecological services, questionnaire development, overestimation of values, and issues of accuracy. However, in some instances, criticisms of stated-preference methods have arisen simply because they are based on behavioral intentions, and they have been scrutinized more carefully than revealed-preference methods, which are based on actual behavior. As the committee has sought to indicate in this chapter and summarized in Table 4-2, both revealed- and stated-preference methods have their advantages and disadvantages, and the choice of method will depend largely on what aquatic ecosystem service is being valued, as well as the policy or management issue that requires valuation.

Lastly, it is important to recognize that each of the economic valuation methods reviewed in this chapter can result in an overestimate or underestimate of individual values for a specific application. Before any empirical study is used in a policy application it is important for the analyst to consider whether the point estimate(s) used underestimate or overestimate the "true" value (see Chapters 6 and 7 for further information).

## APPLICABILITY OF METHODS TO VALUING ECOSYSTEM SERVICES

Given the wide variety of economic methods that are currently available to value aquatic ecosystem services, it may be useful to examine how various methods could be used to value a range of services provided by a single but vitally important aquatic ecosystem. One such ecosystem that has generated several valuation studies of key ecological services is the Great Lakes. The following section reviews these Great Lake studies as an illustration of many of the nonmarket valuation methods and approaches described in this chapter.

### Valuation Case Study: The Great Lakes

The Great Lakes ecosystem covers 94,000 square miles (see also Box 3-2). Collectively, the tributaries to the five Great Lakes drain a territory of 201,000 square miles. Key native species include black bear, bald eagle, wolves, moose, lake trout, and sturgeon, and the lakes surround major migratory flyways for waterfowl, songbirds, and raptors. Thirty-three million people live within the ecosystem and tourism is a major industry year-round. Recreational fishing is annually a multibillion-dollar activity in the regional economy.

In the last 50 years, regional economic changes and pollution control have restored much of the natural beauty of the Great Lakes. However, restoring the ecosystem functions of the Great Lakes remains a priority. Invasive species, such as zebra mussels and lamprey, and exotic fish, such as ruffe and goby, continue to displace and threaten native species. Significant efforts are under way to strengthen populations of Lake Superior walleye, native clams, brook trout, and sturgeon populations.

The ecosystem is also challenged by its industrial history. There are more than 30 areas of concern (AOCs) within the Great Lakes that are burdened with tons of toxic materials (International Joint Commission, 2003). These areas tend to be old industrial areas, harbors, and shipping points. While the mean concentrations tend to be low, these toxic contaminants are typically ingested by small organisms that are in turn successively eaten by other larger organisms. At each stage of the food web, these concentrations become more elevated. The results are excessive (toxic) concentrations of metals and PCBs in fish, waterfowl, and birds of prey. For example, fish consumption advisories for recreational anglers remain in effect in many popular fishing areas across the region.

Like its biological features, the physical character of the Great Lakes ecosystem changes over time. Water levels and volumes have steadily increased over thousands of years (Lewis, 1999), but water levels over the course of decades fluctuate by several feet (Boutin, 2000). The rocky, high shorelines on Lake Superior are fairly stable from a human perspective, but the softer, aggregate and sandy shorelines are susceptible to short-term flooding and long-term erosion. Living in a dynamic ecosystem poses economic risks for managing longer-term investments such as housing, harbor structures, bridges, and roads.

The following three studies address these management issues. The first examines the economic benefits of controlling an exotic species that preys on native fish. The second examines the damages from PCB concentrations in Wisconsin's Fox River, one of the ecosystem's 31 areas of concern. The third explores the economic consequences of ecosystem changes over time.

*Controlling an Exotic Species: Sea Lamprey Invasion*

Sea lampreys are nonnative, eel-like fish that prey on lake trout, sturgeon, salmon, and other large fish in the Great Lakes. Lampreys attach themselves to prey and feed on the bodily liquids of the host fish. The host fish usually dies from infection after the lamprey feeds and detaches. Lamprey were first observed in Lake Ontario in the 1800s and arrived in Lake Michigan by the 1930s (Peeters, 1998).

Lake trout are particularly susceptible to lamprey predation. By the 1950s, lampreys had almost eliminated the self-sustaining lake trout populations in Lake Michigan and Lake Huron (Peeters, 1998). Since the 1950s, vigorous control programs have reduced lamprey populations by 90 percent and led to the restoration of lake trout in Lake Michigan (Great Lakes Fishery Commission, 2002).

However, the lamprey population remains high in Lake Huron. The St. Mary's River is the major uncontrolled spawning area on Lake Huron. The size and volume of the St. Mary's made past control efforts ineffective. Recent improvements in control technology promise much better results at lower costs (Gaden, 1997). An analysis was completed to determine whether the control costs were in line with the recreational fishing benefits of lake trout restoration. The Michigan angling demand model is a statewide travel-cost model of anglers' choices (Hoehn et al., 1996). The model divides the 30-week, non-winter fishing season into 60 fishing choice occasions. Within each occasion, anglers choose whether to go fishing and, if they do, whether they take a day trip or a multiple-day trip. Anglers also choose one of 12 different fishing types, such as cold-water Great Lakes fishing, and fishing location by destination county. Destinations vary in quality by catch rate and other features relevant to fishing choices. In all, the model incorporates 850 distinct choices on each choice occasion.

The model was estimated using a repeated logit statistical framework and data on anglers' choices (Hoehn et al., 1996). The data were obtained from a sample of more than 2,000 Michigan anglers. Sampled anglers were selected randomly from the general population to ensure that the data represented the broad spectrum of Michigan anglers. The sampled anglers were contacted initially at the beginning of the fishing season and then interviewed again (at least) several times over its course. The serial interview approach was used to minimize errors that arise when anglers try to remember a long series of trips. Anglers were also provided with fishing logs to keep track of their trips. Anglers who took frequent trips were interviewed more frequently.

The model estimated the probability of choosing a particular fishing location and type of fishing trip. Trip choices were a function of the distance and travel cost to the location and the quality of fishing. The model was used to estimate benefits for policies that might change fishing quality at a particular site and aggregation of sites, such as inland regions and lakes. For example, an initial analysis indicated that a 10 percent improvement in Lake Michigan and Lake Huron salmon and trout catch rates would result in angler benefits of $3.3 million per year (Lupi and Hoehn, 1998). The analysis considered three alternative ways of controlling lamprey in the St. Mary's River: (1) annual lampricide treatment, (2) annual lampricide and a one-time release of sterile males, and (3) annual lampricide and sterile male release every five years. Treatment costs were several times higher with the third treatment relative to the first, while the trout population and catch rates were only 30 percent higher. Trout populations and catch rates were forecast to increase by 30 to 45 percent in northern Lake Huron and 3 to 7 percent in the central and southern portions of the lake.

The Michigan travel-cost model was used to calculate the benefits of permanent programs of lamprey control using the three different treatments. As the trout population recovers, the third program of continuing lampricide and sterile male releases results in the greatest annual benefits, while the lampricide-only program has the lowest level of annual benefits. However, costs increased with each sterile male release. Although costs increased with treatment, benefits also varied with the geography of catch rate impacts.

Catch rate increases were greatest in the northern region where fewer anglers live and the least in southern Lake Huron nearer the urban areas of Macomb and Wayne Counties in Michigan. As a result, the improvements in catch rates were forecast to occur in areas relatively distant from users. Annual benefits were calculated to be almost twice as large as in the forecast case if the catch rate increase was equal to the same mean but evenly distributed across the entire lake. The result showed that use values decline as the improvement in services was more distant from the users.

The economic outcome of each control alternative was evaluated by examining net benefits. Net benefits were calculated as the present value of benefits minus the present value of costs. Net benefits were positive for each alternative. Using discount rates (see Chapters 2 and 6 for further information) between 3 and 4 percent, net benefits were greatest for annual lampricide and a one-time release of sterile males to quickly reduce the breeding population of lamprey. Net benefits for the first and third alternatives were about the same, meaning that the benefits of continuing sterile male release after the first treatment were just about offset by the costs.

*Fox River Damage from PCBs*

The Fox River enters Green Bay, Wisconsin, on the northwestern shoreline of Lake Michigan. It is the lake's largest tributary. Water, waterpower, and nearby forests supported the early development of the paper industry. By the 1950s, the local paper industry focused on the production of carbonless copy paper. A by-product of its production was the discharge of thousands of pounds of PCBs annually. An estimated 700,000 pounds of PCBs entered the Fox River before PCB use was stopped nationally in 1971. About 20 percent of the PCBs have been deposited in Green Bay and Lake Michigan (Wisconsin DNR, 2001).

Although the human health effects of PCBs are difficult to quantify and measure, the EPA has determined that PCBs cause a range of adverse health effects in animals and that there is "supportive evidence potential carcinogenic and non-carcinogenic effects" in humans (EPA, 2003). To avoid potential adverse health effects in humans, the State of Wisconsin advises anglers to limit their consumption of fish and to prepare fish for consumption so as to avoid fatty tissue that biomagnifies PCBs (Wisconsin DNR, 2001). The primary human use damages are the limitations on eating fish and the increased health risks for anglers and others who choose to eat the fish. Nonuse damages include the impacts on ecosystem functions and other native organisms.

The Wisconsin Department of Natural Resource is conducting a series of studies to estimate economic damages resulting from PCB contamination (Bishop et al., 2000; Breffle et al., 1999; Stratus, 1997). Initial studies focused on injuries to ecosystem functions and services through systematic data collection and analysis (Stratus, 1997). In many cases, it was possible to detect a type of injury but not to quantify its impact on a particular ecosystem service. For instance, PCBs were suspected of injury to fish populations, but it was not possible to quantitatively translate population injuries into estimates of changes in catch rates for sport and subsistence anglers.

The uncertainties regarding service flow injuries led several investigators to two types of damage estimation studies. The first study (Breffle et al., 1999) combined the travel-cost method with stated-preference analysis to estimate use values for anglers. Fishing services to anglers were impaired as a result of both fish consumption advisories (FCAs) and the elevated health risk of eating local fish that FCAs imply. Previous research demonstrated that fishing behaviors change and fishing benefits are reduced by FCAs. The second study (Bishop et al., 2000) used stated-preference analysis to estimate the total values of damages for households in the region. Total value was the sum of both use value damages for anglers and nonuse damages for all households in the study area.

*Pollution Damages to Recreational Fishing*

Breffle et al. (1999) designed a study to estimate the damages to anglers due to FCAs that applied to the Fox River and Green Bay as a result of past PCB releases. Damage estimates were derived from the loss of enjoyment of fishing in an area covered by an FCA and the loss of well-being as a result of fishing at another site, perhaps not covered by an FCA. The study held the number of days of fishing constant at the current, estimated level and did not attempt to estimate damages due to the reduction in the amount of overall fishing.

The analysis estimated the economic demand for fishing as a function of travel cost, whether an FCA was in force at a given site, and other fishing site quality variables. The FCA effect on demand allowed researchers to estimate the shift in fishing demand and the change in consumer value due to presence of the FCA. The reduction in value served as the measure of damages to angling use services.

Data for estimating the demand model were obtained through telephone and mail surveys. The telephone survey used random sample methods to contact a total of 3,190 anglers in northeastern Wisconsin. Respondents were asked to think back over the 1998 angling season and recall their fishing activities. Based on respondents' recollections, the interviewers obtained data on total days spent fishing during 1998, number of days spent fishing in the study area, and attitudes about actions to improve fishing. The mail survey asked respondents to make stated-preference choices across fishing sites that varied in quality. The combined data set allowed researchers to estimate a random utility model of fishing demand conditional on the presence or absence of FCAs in the study area.

The analysis estimated that the 48,600 anglers in the study area fished a total of 641,000 days in 1998. The mean value of damages was \$4.17 per trip (1998 dollars). The present value of fishing use damages was estimated to be \$148 million for a baseline scenario in which natural processes required 100 years to reduce PCBs to levels where FCAs are unnecessary. Restoration efforts that reduced recovery time to 40 years reduced damages to \$123 million, resulting in benefits of \$25 million. Restoration efforts that reduced recovery time to 20 years reduced damages to \$106 million, resulting in cleanup benefits of \$42 million.

*Total Value of Lost Ecosystem Services*

Bishop et al. (2000) investigated the total value of ecosystem services lost due to PCB contamination of the Fox River and Green Bay. That study examined the monetary value of damages as well as the in-kind restoration programs that residents might view as alternatives to removing and containing PCBs. Alternative restoration choices included projects to remove PCB-laden sediments, restore wetlands, enhance recreation, and reduce nonpoint source pollution.

Stated preferences for the restoration alternatives were elicited in a random sample, mail-based survey of 470 households in the study area. The survey questionnaire presented PCB removal as one of several projects to improve natural resources in northeast Wisconsin. The questionnaire also presented six alternative pairs of natural resource programs. Each program within a pair offered different levels of PCB removal, wetland restoration, recreation enhancement, pollution control, and annual tax cost per household.

Respondents were asked to consider each pair and identify their preferred program for each pair. Factorial design methods were used to vary the plans and costs across respondents in the sample. A probit-type discrete choice statistical model was used to estimate the influence of restoration and tax cost on the probability of acceptance. The probit model parameters were then used to calculate willingness to pay a tax cost as a function of the quality of restoration.

The estimates showed that wetlands restoration, improvements in recreational facilities and nonpoint pollution control were poor substitutes for removing and safely containing the PCB-laden sediments. Setting the wetland, recreation, and pollution projects at their maximum levels made up for only 40 years of PCB damages. Natural processes alone were expected to take more than 100 years to reduce PCBs to safe levels.

The present value of PCB damages was estimated to be $610 million (1999 dollars). A restoration that reduced PCBs to safe levels in 40 years resulted in benefits of $248 million by reducing PCB damages to $362 million over the 40-year cleanup interval. An intensive restoration that reduced PCBs to safe levels in 20 years resulted in benefits of $356 million by reducing damages to $254 million over the 20-year cleanup interval.

The final step in the analysis compared the estimated total ecosystem damages with fishing use damages for the 11 percent of households that included at least one angler. This comparison found that estimated total values were 8 to 28 percent greater than use values alone, suggesting that nonuse value was about 8 to 28 percent of use value in angler households.

### *Lakeshore Erosion*

Shoreline erosion offers a short-term laboratory for examining the economic consequences of aquatic ecosystem change. As noted previously, shoreline is valued by property owners for its views, for its proximity to water, and as a location for residential and commercial structures and development. Erosion rates of one to three feet per year do not appreciably affect the amount of shoreline for views, and proximity views and are passed on to the adjacent parcels.

However, erosion does pose a risk of loss of residential and commercial structures, and reducing the risk of loss involves a number of trade-offs. Structures degrade from use and changes in technology in a manner analogous to automobiles and machinery. Locating newly constructed structures far enough away from the existing shoreline so that a building is dilapidated and obsolete before it is threatened by erosion can minimize the risk of erosion to the structure. Increasing the distance to the shore, however, reduces amenities such as panoramic views and increases the time required to get to the beach. Thus, there is a trade-off between the value of these amenities and the economic risk of erosion.

Erosion may be offset for existing structures by physical protection. Rock and concrete armoring protects the shoreline to some extent. However, wave action will eventually undercut such protection. Eroded beaches may sometimes be maintained by dredging offshore sand deposits and using them to replace eroded material. These types of physical protection measures, however, may have impacts on shoreline and coastal ecosystem functions. For instance, armor may reduce erosion of the shoreline, while also reducing sand and sediment flows along the shoreline. Reduced material flows may increase erosion or reduce beach accretion in nearby, unprotected shoreline areas (USACE, 2000).

Economic processes may moderate the risk of erosion to manmade structures by spreading out its consequences over time. In this regard, markets in real property tend to be forward-looking. If there are significant risks from erosion over time, these may be gradually entered into the prices of properties as the risks increase. Buyers are likely to pay more for lower-risk properties and less for higher-risk properties. Property owners may sell a property before the erosion discount becomes higher than the value they place on being near the shore. The annual incremental discount associated with erosion risk might be viewed as part of the cost of a shoreline property, similar to the ordinary costs of depreciation and obsolescence.

Two studies use hedonic methods to examine the impact of erosion risk on the values of shoreline, residential properties. The first examined shoreline property values on Lake Erie (Kriesel et al., 1993), and the second combined data for homes on both Lake Erie and Lake Michigan (Heinz, 2000).

Both studies estimated hedonic regressions where the dependent variable was the logarithm of the sales prices of an individual residential property and the independent variables were the physical characteristics of the property. Physical characteristics included features such as floor area of the structure, parcel size, number of rooms, number of bathrooms, and erosion risk. Erosion risk was measured by the estimated number of years until the shoreline reached the leading, shoreward edge of a structure. The Lake Erie study analyzed data for approximately 300 structures. The combined study used data for 139 structures from the Lake Erie study and data obtained in a mail survey for about 150 Lake Michigan residences.

The results of the two hedonic analyses show that residential property markets are, indeed, forward looking. The major share of erosion's economic cost is incurred long before the actual loss of a residential structure. One way to illustrate this impact uses the estimated hedonic coefficients to calculate the percentage change in property values as years to erosion loss decline. As time to loss declines by 1 year, the property value of a home with a loss in 100 years is discounted by about one-tenth of a percent of its value. At 60 years, a home has lost an accumulated 20 percent of its value due to erosion risk and loses further value at the rate of about 0.6 percent per year. At 20 years, the cumulative discount is 40 percent of the value at 100 years, and the annual discount rate is about 2 percent. At 10 years, the residence is discounted by 60 percent relative to a structure with a risk of 100 years to loss, and the annual rate of loss is 5 percent. At 5 years to loss, a residential structure has lost more than 70 percent of its value relative to the same structure with 100 years to loss.

The analyses show that the cost of erosion is incurred gradually over a long period of time. More than 60 percent of the value of a residence is lost before a residence is within 10 years of the date of its estimated loss. The annual cost of erosion is about $1,400 for a $500,000 residence with an erosion risk of 100 years. For the same structure, the annual cost is about $2,500 at 50 years, $10,400 at 10 years, and $18,400 at 5 years.

## Valuation Case Study: Conclusions

The above studies from the Great Lakes ecosystem illustrate both the strengths and the weaknesses of different valuation methods and approaches. First, the studies show that valuation is a useful tool for assessing a wide range of ecological services and key policy issues concerning management of the Great Lakes, including control of a damaging biological invasion, water pollution by toxic waste, pollution damages to recreational fishing, and the impacts of shoreline erosion. As the extended case study demonstrates, a variety of nonmarket valuation methods are available for assessing these ecosystem management concerns, and if applied correctly, they can yield reliable estimates of the value of key aquatic services. If valuation methods can be applied successfully to a complex and geographically extensive aquatic ecosystem such as the Great Lakes, then nonmarket valuation can also be implemented for equally important aquatic ecosystems elsewhere.

Second, the studies illustrate some of the limitations of revealed- and stated-preference valuation methods discussed earlier in the chapter. For example, the applicability of revealed-preferences methods of valuation depends on whether the ecological service affects peoples' behavior, and whether both the changed environmental condition and the resulting modification in human behavior can be directly or indirectly observed. Thus, for example, the effect of the lamprey invasion could be assessed only in terms of the impact on the recreational fishing benefits of lake trout restoration, which in turn was assessed through the application of a travel-cost model to calculate the possible benefits of alternative lamprey control programs. Clearly, such a valuation estimate can capture only one of many possible complex ecological and economic impacts of the lamprey invasion, although in this instance assessing this recreational benefit was sufficient to determine that the net benefits of lamprey control were positive for all treatments and to identify the preferred treatment method. Similarly, various studies of the health impacts of PCB contamination in the Fox River indicate that the lack of ecological data meant that it was not always possible to quantify how damages to fish populations translate into estimates of changes in catch rates for sport and subsistence anglers, thus limiting reliance on the travel-cost method alone as a method of valuing such impacts. Instead, researchers had to rely either on combined travel-cost and stated-preference methods or on stated-preference methods alone to estimate the total values for households in the region. Although the latter study attempted to separate the households' estimates of use values compared to nonuse values in their overall valuation of the benefits of PCB removal, some of the concerns about the validity of employing contingent valuation techniques to estimate nonuse values may be applicable in this case (NOAA, 1993).

## ISSUES

In describing and discussing currently available nonmarket valuation methods and their applicability to aquatic ecosystem services, a number of key issues have emerged, these include assessing ecological disturbance and threshold effects, limitations to ex ante and ex post valuation, partial versus general equilibrium approaches, and the problem of scope. The following section discusses each of these issues in turn.

### Ecological Disturbance and Threshold Effects

Severe disturbance of an aquatic ecosystem may lead to an abrupt, and possibly very substantial disruption in the supply of one or more ecological services (see Chapter 3 for further information). This "break" in supply is often referred to as a *threshold effect*. The problem for economic valuation is that before the threshold is reached, the marginal benefits associated with a particular ecological service may either be fairly constant or change in a fairly predictable manner with the provision of that service. However, once the threshold is reached, not only may there be a large "jump" in the value of an ecological service, but how the supply of the service changes may be less predictable. Such ecosystem threshold effects pose a considerable challenge, especially for ex ante economic valuation with revealed-preference methods—that is, when one wants to estimate the value of an ecological service that takes into account any potential threshold effects. Since such severe and abrupt changes have not been experienced, peoples' choices in response to them have not been observed. This means that stated-preference methods are the only tool for measuring such values, but there are two complications that warrant discussion.

The first is that there is likely to be considerable uncertainty surrounding both the magnitude and the timing of any threshold effect associated with ecosystem disturbance. Thus, the ecological information may not be available to accurately develop a scenario to describe the ecosystem change in a stated-preference survey. In such a case, a stated-preference survey might be designed to value a variety of plausible ecosystem changes so that it is possible to describe the sensitivity of value estimates to likely outcomes.

The second complication may be that survey respondents will simply reject the valuation scenario as implausible or unbelievable. A large-scale oil spill is one example when survey respondents may reject the valuation scenario out of hand and state that the responsible company should pay for damages, not the general public. Carson et al. (1992) avoided this problem by asking survey respondents to value a public program to prevent an oil spill of the magnitude of the *Exxon Valdez*. Thus, substantial creativity and design effort may be required to develop plausible stated-preference valuation scenarios for large-scale disturbances to aquatic ecosystems that have threshold effects.

Threshold effects can also occur in peoples' preferences. Over some range of change in ecosystem services, marginal values may be quite small, but change dramatically when a drastic change occurs (e.g., listing of an aquatic species as endangered). This suggests that threshold changes in aquatic ecosystem may stimulate threshold changes in preferences. This issue further complicates the valuation of threshold changes because stated-preference valuation methods must be designed to convey the threshold change and motivate people to think how their values would change with the different set of relative prices that would be present after the ecosystem threshold change occurs.

## Limitations of Ex Ante and Ex Post Valuation

The limitations of ex ante valuation using stated-preference methods and real choices are not limited to large-scale, threshold effects. There are many common instances in which people may not have experienced an ecological improvement or degradation and revealed-preference valuation methods are not applicable. Although stated-preference methods are applicable to such changes, it may be difficult for individuals to value trade-offs implied by changes they have not personally experienced. Thus, while stated-preferences are very helpful for ex ante valuation, they are not a complete or infallible solution. There will be circumstances in which nonmarket valuation methods cannot develop accurate value estimates in an ex ante setting.

In the ex post situation, the change has been observed but does not always translate to the revealed choices. For example, the market price of fish may reflect a change in the underlying ecological service, such as the loss of coastal nursery grounds, and thus, there appears to be no value assigned to this ecosystem service. Again, stated-preference methods are the alternative, but they may not be applicable in all situations.

## Partial versus General Equilibrium Approaches

Most valuation methods and valuation studies represent a partial equilibrium approach to a particular policy question. However, as is clear from Chapter 3, the ecological functioning and dynamics that result in most aquatic ecosystem services suggest that to more fully capture the affects of ecosystem changes on the provision of these services, a more general equilibrium approach may be required. A series of independent value estimates for different ecosystem services, when added together, could substantially understate or overstate the full value of changes in all services. The key issue is whether there is substitute or complementary relationships between the services (Hoehn and Loomis, 1993).

As discussed above, there have been a number of recent attempts to use such an approach, or integrated economic-ecological modeling, to value various services of aquatic ecosystems. In essence, these approaches represent the extension of the production function approach to a full ecosystem level.

## Scope

Insensitivity to scope is a major issue in contingent valuation studies of nonuse values of ecosystem services. This issue was raised by the National Oceanic and Atmospheric Administration Panel on Contingent Valuation (1993), which stated that this problem demonstrates "inconsistency with rational choice." Insensitivity to scope is exhibited by value estimates' being insensitive to the magnitude of the ecosystems change being valued. For example, if values estimated for restoring 100 and 1,000 acres of wetlands were statistically identical, this would indicate lack of sensitivity to scope. The inconsistency with rational choice arises because it is expected that people would pay more for the larger restoration project, all other factors being equal. The basis for the NOAA panel's concern was a study by Boyle et al. (1994) who found that estimates of nonuse values were not sensitive to whether 2,000, 20,000, or 200,000 bird deaths were prevented in waste oil holding ponds. While this study was criticized in a variety of public fora, Ahearn et al. (2004) reported a similar result in another study of grassland bird numbers. Notably, this latter study generally followed the NOAA panel's (1993) guidelines for the design of a credible contingent valuation study of nonuse values.

Insensitivity to scope is a major issue for valuing aquatic ecosystems services because stated-preference methods, which include contingent valuation, are likely to be important in estimating many component values in a TEV framework. There are many instances in which there is no visible behavior that supports the use of revealed-preference methods, although two important caveats should be considered.

First, the NOAA panel focused on the use of contingent valuation to estimate nonuse values. There will be many cases in which stated-preference methods are needed to estimate use values for aquatic ecosystem services. Sensitivity to scope has been demonstrated clearly in the estimation of use values in the literature, and some of these studies are applications to aquatic ecosystems (e.g., Boyle et al., 1993). In fact, Carson (1997) provides a list of contingent valuation studies that have demonstrated scope effects when use values are involved, and the vast majority of these studies have implications for valuing aquatic ecosystem services. Moreover, Carson et al. (1996) show that contingent valuation estimates are comparable to similar revealed-preference estimates—thereby, demonstrating the convergent validity of the stated-preference and revealed-preference estimates. Thus, the literature supports the use of contingent valuation for estimating use values for aquatic ecosystem services.

The second caveat applies to the use of contingent valuation to estimate nonuse values. Although the NOAA panel stated that contingent valuation can provide useful information on nonuse values, the ability of contingent valuation methods to demonstrate scope effects has not been demonstrated clearly in the literature. This a major concern for valuing aquatic ecosystems because nonuse values would be expected to be an important and large component of any total economic value assessment. In this regard, attribute-based, conjoint analysis provides a promising option. This approach presents the description of the aquatic ecosystem to be valued in component services and clearly informs survey respondents that there are different levels of these services. Respondents are then asked to select alternatives that differ in terms of the component services. This relative context has been shown to demonstrate scope effects (Boyle et al., 2001). The key difference is that contingent valuation has used a between-subjects design where independent samples are asked to value each of the different levels of the ecosystem. Conjoint analysis uses a within-subjects design where each respondent sees multiple levels of the ecosystem. Although a between-subjects design is appealing from an experimental design perspective, this is not the way real-world decisions are made. People make revealed choices where they observe ecosystem goods and services with different levels of attributes, and whereas conjoint analysis mimics this choice framework, contingent valuation does not. A question then arises as to what standard should contingent valuation be held. A between-subjects design to test for scope holds contingent valuation to a higher standard than market decisions are based upon (Randall and Hoehn, 1996), whereas the within-subject design of conjoint analysis mimics the relative choices that occur in markets. These results imply that conjoint analysis may be the better method to employ in estimating nonuse values for aquatic ecosystems services.

## SUMMARY: CONCLUSIONS AND RECOMMENDATIONS

This chapter demonstrated that there is a variety of nonmarket valuation approaches that can be applied to valuing aquatic and related terrestrial ecosystem services.

For revealed-preference methods, the types of applications are limited to a set number of specific aquatic ecosystem services. However, both the range and the number of services that can potentially be valued are increasing with the development of new methods, such as dynamic production function approaches, general equilibrium modeling of integrated ecological-economic systems, conjoint analysis, and combined revealed- and stated-preference approaches.

Stated-preference methods can be applied more widely, and certain values can be estimated only through the application of such techniques. On the other hand, the credibility of estimated values for ecosystem services derived from stated-preference methods has often been criticized in the literature. For example, contingent valuation methods have come under such scrutiny that it led to the NOAA panel guidelines of "good practice" for these methods.

Benefit transfers and replacement cost/cost of treatment methods are increasingly being used in environmental valuation, although their application to aquatic ecosystem services is still limited. Economists generally consider benefit transfers to be a "second-best" valuation method and have devised guidelines governing their use. In contrast, replacement cost and cost of treatment methods should be used with great caution if at all. Although economists have attempted to design strict guidelines for using replacement cost as a last resort "proxy" valuation estimation for an ecological service, in practice estimates employing the replacement cost or cost of treatment approach rarely conform to the conditions outlined by such guidelines.

Although the focus of this chapter has been on presenting the array of valuation methods and approaches currently available for estimating monetary values of aquatic and related terrestrial ecosystem services, it is important to remember that the purpose of such valuation is to aid decision-making and the effective management of these ecosystems. Building on this critical point, at least three basic questions arise for any method that is chosen to value aquatic ecosystem services:

1. Are the services that have been valued those that are the most important for supporting environmental decision-making and policy analyses involving benefit-cost analysis, regulatory impact analysis, legal judgments, and so on?
2. Can the services of the aquatic ecosystem that are valued be linked in some substantial way to changes in the functioning of the system?
3. Are there important services provided by aquatic ecosystems that have not yet been valued so that they are not being given full consideration in policy decisions that affect the quantity and quality of these systems?

In many ways, the answers to these questions are the most important criteria for judging the overall validity of the valuation method chosen.

It is clear that economists and ecologists should work together to develop valid estimates of the values of various aquatic ecosystem services that are useful to inform policy decision-making. The committee's assessment of the literature is that this has not been done adequately in the past and most valuation studies appear to have been designed and implemented without any such collaboration. Chapter 5 helps to begin to build this bridge.

The range of ecosystem services that have been valued to date are very limited, and effective treatment of aquatic ecosystem services in benefit-cost analyses requires that more services be subject to valuation. Chapter 3 begins to develop this broad perspective of aquatic ecosystem services.

Nonuse values require special consideration; these may be the largest component of total economic value for aquatic ecosystem services. Unfortunately, nonuse values can be estimated only with stated-preference methods, and this is the application in which these methods have been soundly criticized. This is a clear mandate for improved valuation study designs and more validity research.

There is a variety of nonmarket valuation methods that are available and presented in this chapter. However, no single method can be considered the best at all times and for all types of aquatic ecosystem valuation applications. In each application it is necessary to consider what method(s) is the most appropriate.

In presenting the various nonmarket valuation methods available for estimating monetary values of aquatic and related terrestrial ecosystem services, this chapter has also sought to provide some guidance on the appropriateness of the various methods available for a range of different services. Based on this review of the current literature and the preceding conclusions, the committee makes the following recommendations:

- There should be greater funding for economists and ecologists to work together to develop estimates of the monetary value of the services of aquatic and related terrestrial ecosystems that are important in policymaking.

- Specific attention should be given to funding research at the "cutting edge" of the valuation field, such as dynamic production function approaches, general equilibrium modeling of integrated ecological-economic systems, conjoint analysis, and combined stated-preference and revealed-preference methods.
- Specific attention should be given to funding research on improved valuation study designs and validity tests for stated-preference methods applied to determine the nonuse values associated with aquatic and related terrestrial ecosystem services.
- Benefit transfers should be considered a "second-best" method of ecosystem services valuation and should be used with caution, and only if appropriate guidelines are followed.
- The replacement cost method and estimates of the cost of treatment are not valid approaches to determining benefits and should not be employed to value aquatic ecosystem services. In the absence of any information on benefits, and under strict guidelines, treatment costs could help determine cost-effective policy action.

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# 5

# Translating Ecosystem Functions to the Value of Ecosystem Services: Case Studies

## INTRODUCTION

Valuing ecosystem services requires the integration of ecology and economics. Ecology is needed to comprehend ecosystem structure and functions and how these functions change with different conditions. Both ecology and economics are required to translate ecosystem functions into the production of ecosystem goods and services. Economics is needed to comprehend how ecosystem goods and services translate into value (i.e., benefits for people; see also Figure 1-3). The two preceding chapters discuss much of the relevant ecological and economic literature. Chapter 3 focuses on the relevant ecological literature on aquatic and related terrestrial ecosystem functions and services, while Chapter 4 focuses on the economic literature on nonmarket valuation methods useful for valuing ecosystem goods and services. In this chapter, the focus is on the integration of ecology and economics necessary for valuing ecosystem services for aquatic and related terrestrial ecosystems. More specifically, a series of case studies is reviewed (including those taken from the eastern and western United States; see Chapter 1 and Box ES-1 for further information), ranging from studies of the value of single ecosystem services, to multiple ecosystem services, to ambitious studies that attempt to value all services provided by ecosystems. An extensive discussion of implications and lessons learned from these case studies is provided and precedes the chapter summary.

Development of the concept of ecosystem services is relatively recent. Only in the last decade have ecologists and economists begun to define ecosystem services and attempted to measure the value of these services (see for example, Balvanera et al., 2001; Chichilnisky and Heal, 1998; Constanza et al., 1997; Daily, 1997; Daily et al., 2000; Heal, 2000a,b; Pritchard et al., 2000; Wilson and Carpenter, 1999). There is a much longer history of natural resource managers and economists evaluating "goods" produced by ecosystems (e.g., forest products, fish production, agricultural production). For example, in 1926, Percy Viosca, Jr., a fisheries biologist, estimated that the value of conserving wetlands in Louisiana for fishing, trapping, and collecting activities was $20 million annually (Vileisis, 1997). In the 1960s and early 1970s, pioneering work by Krutilla (1967), Hammack and Brown (1974), and Krutilla and Fisher (1975), among others, greatly expanded the set of "goods and services" generated by natural systems considered by economists to be of value to humans (e.g., clean air, clean water, recreation, ecotourism). Economic geographers and regional scientists (e.g., Isard et al., 1969) examined spatial relationships among natural and socioeconomic systems. Recent work on ecosystem services has broadened the set of goods and services studied to include water purification, nutrient retention, and flood control, among other things. It has also emphasized the importance of understanding natural processes within ecosystems (e.g., primary and secondary productivity, carbon and nutrient cycling, energy flow) in order to understand the production of ecosystem services. Yet, as discussed throughout this report, for the most part, the importance of these natural processes in producing ecosystem services on which people depend has remained largely invisible to decision-makers and the general public. For most ecosystem services, there are no markets and no readily observable prices, and most people are unaware of their economic value. All too often it is the case that the value of ecosystem services becomes apparent only after such services are diminished or lost, which occurs once the natural processes supporting the production of these services have been sufficiently degraded. For example, the economic importance of protecting coastal marshes that serve as breeding grounds for fish may become apparent only after commercial fish harvests decline. By then, it may be difficult or impossible to repair the damage and restore the production of such services.

Although there has been great progress in ecology in understanding ecosystem processes and functions, and in economics in developing and applying nonmarket valuation techniques for their subsequent valuation, at present there often remains a gap between the two. There has been mutual recognition among at least some ecologists and some economists that addressing issues such as conserving ecosystems and biodiversity requires the input of both disciplines to be successful (Daily et al., 2000; Kinzig et al., 2000; Loomis et al., 2000; Turner et al., 2003; Holmes et al., 2004). Yet there are few existing examples of studies that have successfully translated knowledge of ecosystems into a form in which economic valuation can be applied in a meaningful way (Polasky, 2002). Several factors contribute to this ongoing lack of integration. First, some ecologists and economists have held vastly different views on the current state of the world and the direction in which it is headed (see, for example, Tierney, 1990, who chronicles the debates between a noted ecologist and economist [Paul Ehrlich and Julian Simon]). Second, ecology and economics are separate disciplines, one in natural science and the other in social science. Traditionally, the academic organization and reward structure for scientists make collaboration across disciplinary boundaries difficult even when the desire to do so exists. Third, as noted previously, the concept of ecosystem services and attempts to value them are still relatively new. Building the necessary working relationships and integrating methods across disciplines will take time.

Some useful integrated studies of the value of aquatic and related terrestrial ecosystem goods and services are starting to emerge. The following section reviews several such studies and the types of evaluation methods used. This review begins with situations in which the focus is on valuing a single ecosystem service. Typically in these cases, the service is well defined, there is reasonably good ecological understanding of how the service is produced, and there is reasonably good economic understanding of how to value the service. Even when valuing a single ecosystem service however, there can be significant uncertainty about either the production of the ecosystem service, the value of the ecosystem service, or both. Next reviewed are attempts to value multiple ecosystem services. Because ecosystems produce a range of services that are frequently closely connected, it is often difficult to discuss the valuation of a single service in isolation. However, valuing multiple ecosystem services typically multiplies the difficulty of valuing a single ecosystem service. Last to be reviewed are analyses that attempt to encompass all services produced by an ecosystem. Such cases can arise with natural resource damage assessment, where a dollar value estimate of total damages is required, or with ecosystem restoration efforts. Such efforts will typically face large gaps in understanding and information in both ecology and economics.

Proceeding from single services to entire ecosystems illustrates the range of circumstances and methods for valuing ecosystem goods and services. In some cases, it may be possible to generate relatively precise estimates of value. In other cases, all that may be possible is a rough categorization (e.g., "a lot" versus "a little"). Whether there is sufficient information for the valuation of ecosystem services to be of use in environmental decision-making depends on the circumstances and the policy question or decision at hand (see Chapters 2 and 6 for further information). In a few instances, a rough estimate may be sufficient to decide that one option is preferable to another. Tougher decisions will typically require more refined understanding of the issues at stake. This progression from situations with relatively complete to relatively incomplete information also demonstrates what gaps in knowledge may exist and the consequences of those gaps. Part of the value of going through an ecosystem services evaluation is to identify the gaps in existing information to show what types of research are needed.

## MAPPING ECOSYSTEM FUNCTIONS TO THE VALUE OF ECOSYSTEM SERVICES: CASE STUDIES

Despite recent efforts of ecologists and economists to resolve many types of challenges to successfully estimating the value of ecosystem services, the number of well-studied and quantified cases studies remains relatively low. The following section reviews cases studies that have attempted to value ecosystem services in the context of aquatic ecosystems. These examples illustrate different levels of information and insights that have been gained thus far from the combined approaches of ecology and economics.

### Valuing a Single Ecosystem Service

This review begins with studies of the value of ecosystem services using examples that attempt to value a single ecosystem service. These cases provide the best examples of both well-defined and quantifiable ecosystem services and of services that are amenable to application of economic valuation methodologies. The best-known example of a policy decision hinging on the value of a single ecosystem service involves the provision of clean drinking water for New York City, which is reviewed first. Other examples include cases where ecosystems provide habitat for harvested fish or game species and cases where they provide flood control.

In all of the cases reviewed in this section, the ecosystem service is well-defined although there may be some scientific uncertainty surrounding quantification of the amount of the service provided. In some cases, adequate methods for valuing the single ecosystem service exist. Further, for some cases, such as the New York City example below, information about a single ecosystem service may prove sufficient to support rational environmental decision-making. In other cases, this will not be so, and further work to assess a more complete set of ecosystem services will be necessary. Under no circumstances, however, should the value of a single ecosystem service be confused with the value of the entire ecosystem, which has far more than a single dimension. Unless it is kept clearly in mind that valuing a single ecosystem service represents only a partial valuation of the natural processes in an ecosystem, such single service valuation exercises may provide a false signal of the total economic value of the natural processes in an ecosystem.

*Providing Clean Drinking Water: The Catskill Mountains and New York City's Watershed*

One of the best-studied water supply systems in the world is the one that provides drinking water for more than 9 million people in the New York City metropolitan area (Ashendorff et al., 1997; NRC, 2000a; Schneiderman, 2000). New York City's water supply includes three large reservoir systems (Croton, Catskill, and Delaware) that contain 19 reservoirs and 3 controlled lakes. This system, including all tributaries, encompasses a total area of 5,000 $km^2$ with a reservoir capacity of $2.2 \times 10^9$ $m.^3$ This complex array of natural watersheds requires a wide range of management to sustain the water quality supplied to the reservoirs and aqueducts. Historically, these watersheds have supplied high-quality water with little contamination. However, increased housing developments with septic systems, combined with nonpoint sources of pollution such as runoff from roads and agriculture, have posed threats to water quality. Further significant deterioration of water quality would force U.S. Environmental Protection Agency (EPA) to require New York City to build a water filtration system[1] to ensure that drinking water delivered to consumers would meet federal drinking water standards. By 1996, New York City faced a choice: it could either build water filtration system or protect its watersheds to ensure high-quality drinking water.

The cost of building new, larger filtration system necessary to meet water quality standards was estimated to lie in the range of $2 billion to $6 billion. Moreover, the city estimated that it would spend $300 million annually to operate the new filtration plant. Together, the costs of building and operating the filtration system were estimated to be in the range of $6 billion to $8 billion (Chichilnisky and Heal, 1998).

Instead of investing in a water filtration facility, New York City opted to invest more in protecting watersheds. Maintaining water quality in the face of increased human population densities in the watershed required increased protection of riparian buffer zones along rivers and around reservoirs. These zones help to regulate nonpoint sources of nutrients and pesticides from stormwater runoff, septic tanks, and agricultural sources. In 1997 the city received "filtration avoidance status" from the EPA by promising to upgrade watershed protection. The 1997 Watershed Memorandum Agreement resulted from negotiations among the State of New York, New York City, the EPA, municipalities within the watershed, and five regional environmental groups. The agreement provided a framework for compliance with water quality standards and contained plans for land acquisition through mutual consent, watershed regulations, environmental education workshops, and partnership programs with community groups. For example, a farmer-led Watershed Agricultural Council provides programs for the approximately 350 dairy and livestock farms in the watershed to minimize nutrient input from agricultural runoff (Ashendorff et al., 1997).

Under this agreement, New York City is obligated to spend $250 million during a 10-year period to purchase lands within the watershed (up to 141,645 hectares). In this part of the overall response, the New York City Department of Environmental Protection land acquisition program purchases undeveloped land from willing sellers rather than relying on condemnation and the power of eminent domain. Property rights to develop land in the watershed rests in the hands of local landowners. In some cases these rights are regulated by local ordinances. New

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[1] In the late 1990s, the plan was to build one centralized plant for the Catskill/Delaware portion of the larger watershed (see NRC, 2000a for further information). However, it has since been determined that the Croton portion of the watershed has to build a separate filtration plant.

York City's 1953 Watershed Rules and Regulations give the city some authority over watershed development to limit water pollution. Decades-old resentment remains among some residents of upstate watersheds because earlier land acquisitions to build the reservoirs displaced entire communities. Moreover, recent concerns about security of the reservoirs have also polarized residents whose road access has been limited. Exactly what legal rights New York City has and what legal rights local municipalities and local landowners have to make decisions is not fully resolved. The long-term costs of riverbank protection, upkeep of sewage treatment plants by municipalities and overall maintenance costs of this approach remain uncertain.

On the other hand, a series of regulations prohibiting certain types of development in certain places (e.g., areas in close proximity to watercourses, reservoirs, reservoir stems, controlled lakes, wetlands) was agreed upon. The city together with the Catskill Watershed Corporation developed a comprehensive geographical information system to track land uses and to analyze runoff and storm flows resulting from precipitation. Runoff is sensitive to connections among stream network, and to the amount of impervious surface in the watershed (e.g., roads, buildings, driveways, parking lots), which results in increased peak flows that can cause flooding and bank erosion (Arnold and Gibbons, 1996; Gergel et al., 2002). To minimize these effects, new construction of impervious surfaces within 300 feet of a reservoir, rivers, or wetland is prohibited. Road construction within 100 feet of a perennial stream and 50 feet of an intermittent stream is also prohibited. Septic system fields cannot be located within 100 ft of a wetland or watercourse or 300 feet of a reservoir because these on-site sewage treatment and disposal systems do not work effectively in saturated soil. Septic fields also interfere with the natural nutrient processing in floodplains, wetlands, and riparian buffer zones along streams. Funds are available to subsidize upgrades of local wastewater treatment plants and septic systems throughout the watershed. There are 38 wastewater treatment plants in the watershed that are not owned by New York City. Overall, New York City projected that it would invest $1 billion to $1.5 billion in protecting and restoring natural ecosystem processes in the watershed (Ashendorff et al., 1998; Chichilnisky and Heal, 1998; Foran et al., 2000; NRC, 2000a). Incentives for landowners to improve riparian protection through conservation easements and educational outreach efforts were combined with management of state-owned lands to minimize erosion and protect riparian buffers.

In this case, it was not necessary to value all or part of the services of the Catskills watershed; it was merely necessary to establish that protecting and restoring the ecological integrity of the watershed to provide clean drinking water was less costly than replacing this ecosystem service with a new water filtration plant. As discussed in Chapter 4, Shabman and Batie (1978) suggest that a replacement cost approach can provide a "proxy" valuation estimation for an ecological service if the alternative considered provides the same service, the alternative compared is the least-cost alternative, and there is substantial evidence that the service would be demanded by society if it were provided by that least-cost alternative. In the Catskills case the proposed filtration plant would provide very similar services (more on this below). Of course, the city will have to provide clean water somehow. So these conditions are met and the cost of replacing the provision of clean drinking provided by the watershed with a filtration plant, less the cost of protecting and restoring the watershed, can be thought of as a measure of the ecosystem service value to New York City as a water purification tool. If, however, demand side management can reduce demand for water at less cost than it costs to provide the water via the filtration plant, then demand side management costs would provide the relevant avoided costs. Both methods, natural processes in watersheds and a water filtration plant, are capable of providing clean drinking water that meets drinking water standards.

This case also appears to provide clear environmental policy direction. For New York City, it is likely to be far less costly to provide safe drinking water by protecting watersheds, thereby maintaining natural processes, than to build and operate a filtration plant. Further, protecting watersheds to provide clean water also enhances provision of other ecosystem services (e.g., open space for recreation, habitat for aquatic and terrestrial species, aesthetics). As discussed throughout this report, such ecosystem services are arguably far harder to value economically. Since these values add to the value of protecting watersheds for the provision of clean water, which is the preferred option even without consideration of these additional values, it is not necessary to establish a value for these services for policy purposes. Thus, protecting watersheds can be justified on the basis of the provision of clean drinking water alone.

Despite the appearance of being a textbook case for valuing a single ecosystem service, several issues make the answer to ecosystem valuation less obvious than at first glance. The replacement cost approach assumes that the same service will be provided under either alternative. In reality, it is unlikely that watershed protection and filtration will provide identical levels of water quality and reliability over time because engineered systems can fail—especially during storms when heavy flows overwhelm the system. Likewise, natural watersheds can also vary in their effectiveness in response to severe storm flows or other disturbances (Ashendorff et al., 1997). Managed watersheds can require some maintenance costs to sustain ecosystem services such as clean up of accidental spills or fish kills to prevent pollution or control of invasive species such as zebra mussels (Covich et al., 2004; Giller et al., 2004). Both engineered and ecosystem approaches are vulnerable but they differ in the types of uncertainty associated with each investment.

New York City's watershed investment plan includes several maintenance costs such as thorough, multistaged monitoring of water quality and disease surveillance that triggers active management and localized water treatment. Baseline data on water quality and biodiversity of stream organisms in the watershed (e.g., aquatic insects) are being collected by the Stroud Water Research Center (2001) annually to determine if the city's recent management efforts are effective. By reducing the risk of contaminants from various sources, the city can minimize use of disinfectants at the final water treatment stages. Reducing chemical use saves money directly and it may also have health benefits since chlorination can produce halogenated disinfection by-products (e.g., chloroform, trihalomethane) in drinking water, especially in ecosystems with high levels of organic matter (Symanski et al., 2004; Villanueva et al., 2001; Zhang and Minear, 2002). Some of these by-products may be carcinogens. On the other hand, filtration may provide higher-quality drinking water because chlorination is not completely effective in killing pathogens, particularly when there are high levels of suspended materials (Schoenen, 2002).

Despite the regulations and the comprehensive framework contained in the city's watershed protection plan, considerable uncertainties exist about whether the plan can sustain high quality water supplies over the longer-term. Enforcement of the regulations and monitoring the rapid rate of suburban growth constitute a major challenge, and these development pressures in the area may increase the opportunity costs of watershed protection. Construction in the headwaters of streams, permitted under the plan, may result in increased runoff rates and erosion. Filling tributary channels with sediments can take place incrementally, with each step occurring at a small scale. Yet numerous small-scale changes may transform the watershed in detrimental ways over time without sufficient oversight and long-term planning. The U.S. Army Corps of

Engineers (USACE) has authority under Section 404 of the Clean Water Act to review permits. However, without site-by-site reviews of small projects (less than four hectares), allowable incremental alterations can have significant cumulative effects on small streams. Decreased stream density (stream length per drainage basin area) would occur if natural stream channels were replaced by pipes and paved over for development, resulting in loss of the essential ecological processes of organic matter breakdown and sediment retention (Meyer and Wallace, 2001; Paul and Meyer, 2001).

Additional uncertainties might impact decision-making, besides the adequacy of protection in the watersheds. Model uncertainty that arises from imperfect understanding of ecosystem function and the translation to ecosystem services is a major issue for most ecosystem valuation studies. In this case, there is model uncertainty because the hydrologic modeling used for determining water supplies is affected by the definition of spatial and temporal boundaries. For example, other municipalities in New York and New Jersey use water from the Catskills. Changes in water diversions from the Catskill Mountains can affect outflows to the Delaware River and modify salinities in the lower sections of the river used by Philadelphia (Frei et al., 2002). Given the additional uncertainties of future regional droughts, floods, and extreme temperatures, as well as acid rain and nitrogen deposition from atmospheric sources, planners must consider the range of intrinsic natural variability in decision-making. Planners can cope with aspects of model and parameter uncertainty by carefully monitoring land uses in the basin and incorporating environmental data into any new regulations that might be required. A long series of studies on nutrient budgets and acid deposition provides some essential baseline information for the Catskills (e.g., Frei et al., 2002; Lovett et al., 2000; Murdoch and Stoddard, 1992, 1993; Stoddard, 1994). Other locations may lack sufficient information, and thus, considerable sources of uncertainty will limit the analysis of complete replacement costs.

In this case, the provision of clean drinking water supplies through the protection of natural processes in watersheds rather than through the human-engineered solution of building a water filtration system offers an estimate of the value of restoring an ecosystem service that provides clear advice to a policy decision. Replacement costs for natural processes in watersheds providing clean drinking water are estimated to be in the neighborhood of $6 billion to $8 billion, which is far higher than estimates of the cost necessary to protect the watersheds. Because the policy question is relatively specific (i.e., whether to build a filtration plant or to protect watersheds), currently available economic methods of ecosystem service valuation are sufficient.

Even in this example however, obtaining a precise estimate of the value of the provision of clean water through watershed conservation is probably not possible given existing knowledge. First, it is not clear that the two methods, filtration and watershed protection, provide the same level of water quality and reliability. There are numerous dimensions to the provision of clean drinking water, such as the concentrations of various trace chemicals, carcinogens, and suspended solids, variance of the quality, and the adequacy of supply. It is unlikely that the two methods will deliver water that is identical in all of these dimensions under all conditions. Second, there is no guarantee that protecting watersheds will continue to be successful. Increased development pressure on lands outside the riparian buffer zones or inadequate enforcement may require building a filtration system at some point in the future. If the watershed protection plans prove to be insufficient in the future, the investments in protection will still likely reduce future costs of building filtration plants because the quality of the water to be treated will be enhanced through these land-use programs.

Finally, it should be emphasized that (1) the value of providing clean drinking water is only a partial measure of the value of ecosystem services provided by the watershed, and (2) replacement cost is rarely a good measure of the value of an ecosystem service. Even if water quality benefits alone did not justify watershed protection, such a finding would not justify abandoning efforts at watershed protection. To make that decision would require a broader effort to measure the value of the wider set of ecosystem services produced by Catskills watersheds. It is less clear that estimates to answer this broader question are sufficiently precise to provide policy-relevant answers (see Chapters 2 and 6 for more on framing). Replacement cost methods can be used as a measure of the value of ecosystem services only when there are alternative ways to provide the same service and when the service will be demanded if provided by the least cost alternative. Replacement cost does not constitute an estimate of value of the service to society. It represents the value of having the ability to produce the service through an ecosystem rather than through an alternative method.

### *Other Surface Water Examples*

Other cities have used similar strategies to invest in maintaining the ecological integrity of their watersheds as a means of providing high quality drinking water that meets all federal, state, and local standards. Boston, Seattle, San Francisco, and Greenville, South Carolina, are other examples where the value of ecosystem services could be estimated using a replacement cost approach for building and operating water treatment plants that are roughly equivalent in the quality of drinking water supplied (NRC, 2000a). The costs of producing safe drinking water were traditionally derived from production cost estimates associated with engineering treatments. Filtration plants were built to remove organic materials, and then some form of chemical purification was used to control microorganisms. Engineers generally considered natural ecosystems such as rivers and lakes mostly from the viewpoint of volumes, transport systems, resident times, dilution, and natural "reoxygenation." In other words, they viewed many natural ecosystems as large pipes rather than as complex habitats for a diverse biota. Yet even viewed strictly through the lens of water supply systems, protecting natural processes within ecosystems may be superior to engineering solutions, and such a result may be sufficient for decision-making purposes. Replacement cost estimates for provision of clean drinking water, however, provide an estimate of just one source of value and should not be confused with the complete value of ecosystem services provided by watersheds. Further, as discussed in Chapter 4, replacement cost is a valid approach to economic valuation only in highly restricted circumstances—namely, that there are multiple ways to achieve the same end and the benefits exceed the costs of providing this end.

### *Provision of Drinking Water from Groundwater: San Antonio, Texas*

In contrast with the Catskills case, there has been a lack of valuation studies to date on the economic value of the Edwards Aquifer (see also Box 3-5) that supplies drinking water to San Antonio as well as water for irrigation and other uses. Groundwater supplies approximately half of America's drinking water (EPA, 1999). It is relied on heavily in some parts of the arid West where surface waters are scarce. The long-term supply of groundwater is a concern in some of these areas (Howe, 2002; Winter, 2001). For example, depletion of the Ogallala Aquifer is creating great uncertainties about future water supplies throughout a large region of the central United States (Glennon, 2002; Opie, 1993). Similarly, depletion of groundwater aquifers in the Middle Rio Grande Basin is creating uncertainty about the future supply of drinking water for Albuquerque, New Mexico (NRC, 1997, 2000b). Aquifers generally provide high quality drinking water, but pollution lowers water quality in some areas, such as the Cape Cod Aquifer where there are threats from sewage and toxic substances leaching into groundwater from the Massachusetts Military Reservation (Barber, 1994; Morganwalp and Buxton, 1999).

The long-term sustainability of groundwater depends on matching extraction with recharge (Sanford, 2002). It is often difficult to predict the timing and rate of recharge because of complications of local geology, time lags, and climate uncertainties. Recharge of the porous karstic limestone that characterizes the Edward Aquifer occurs primarily during wet years when precipitation infiltrates deeply into the soils and underlying rock (Abbott, 1975). Drought conditions have complex effects on lowering recharge rates while simultaneously tending to increase the demand for water. The greatest source of uncertainty about groundwater recharge is the range of natural interannual variability in precipitation and land-use changes. Increasing demands from a growing population and the difficulty in predicting climate change raise questions about the adequacy of groundwater supplies in arid regions (Grimm et al., 1997; Hurd et al., 1999; Meyer et al., 1999; Murdoch et al., 2000).

Aquifer depletion has both economic and ecological consequences. The costs for deeper drilling and pumping increase as groundwater is depleted. Removal of water in the underground area may cause collapse of the overlying substrata. These collapses decrease future storage capacity below ground and may cause damage on the surface as areas subside, buckle, or collapse. In some areas, depleted groundwater may cause the intrusion of low-quality water from other aquifers or from marine-derived salt or brackish waters that could not readily be restored for freshwater storage and use.

Depletion of groundwater supplies creates uncertainty and generally is offset by supplies from surface waters. An interesting exception is San Antonio (the ninth largest city in the United States) that relies primarily on groundwater for its source of municipal water. An outbreak of cholera in 1866 from polluted surface waters prompted the City of San Antonio to switch to groundwater from the Edwards Aquifer. The aquifer is estimated to contain up to 250 million acre-feet of water with a drainage area covering approximately 8,000 square miles. The average annual recharge is estimated at approximately 600,000 acre-feet of water (Merrifield , 2000). Given this large supply, the Edwards Aquifer plays a major role in the economy of San Antonio and south-central Texas (Glennon, 2002). In some parts of this region, clean, free-flowing springs and artesian wells provide drinking water without the cost of pumping and with minimal treatment. San Antonio built its first pumping station in 1878. The U.S. Geological Survey (USGS) has monitored aquifer recharge rates since 1915 and water quality monitoring began in 1930. In 1970 the Edwards Aquifer was designated a "sole source aquifer" by the EPA under the Safe Drinking Water Act. Currently, more than 1.7 million people rely on the Edwards Aquifer for water. Industrial and agricultural demands on the Edwards Aquifer have increased, and the city has planned for new reservoir storage as part of its water supply several times over the last two decades. As the demand for water in the area has grown, concerns have arisen over both the quantity and the quality of groundwater available (Wimberley, 2001).

Depletion also raises the specter that adequate supply will not be available for future demand at any price. The $3.5 billion-a-year tourist industry in San Antonio is centered on the city's River Walk, which relies primarily on recycled groundwater (Glennon, 2002). Uncertainties over the long-term availability of water make long-term planning problematic and threaten long-term investments. For example, aquaculture companies (e.g., Living Waters Artesian Springs, Ltd.) expanded their catfish operations in March 1991, but subsequently closed in November 1991 because of concerns over pumping rates and the impaired water quality of return flows (i.e., high concentrations of dissolved nutrients) to surface- and groundwaters associated with the Edwards Aquifer.

Groundwater storage is critical in most aquatic ecosystems to provide persistence spring and stream habitats during dry seasons or during drought. Several springs (Comal, San Antonio, San Pedro) in the area began to dry up following a seven-year drought in the 1950s. Chen et al. (2001) used a climate change model to estimate the regional loss of welfare at $2.2 million to $6.8 million per year from prolonged drought. They estimated groundwater recharge based on historic data for recharge rates as influenced by precipitation and temperature. These researchers forecasted municipal and irrigation demand for five scenarios, including current condition and four different levels of climate change. Estimates of demand elasticity were based on models and methods used in other studies of arid regions. Given the projected reductions in available water, it would be necessary to protect endangered species in springs and groundwater, at an additional reduction of 9 to 20 percent in pumping that would add $0.5 million to $2 million in costs.

The economic value of organisms living in groundwater and in springs, wetlands, and downstream surface flows supplied by groundwater is difficult to estimate. However, their value is generally assumed to be high because of their many functional roles in maintaining clean water as well as their existence values. For example, many diverse microbial communities and a wide range of invertebrate and vertebrate species live in groundwater, springs, and streams (Covich, 1993; Gibert et al., 1994; Jones and Mulholland, 2000). Their main functions are breaking down and recycling organic matter that forms the base of a complex food web (Covich et al., 1999, 2004). Depletion of groundwater aquifers results in possible loss of habitat for endemic species protected by state and federal regulations. For example, the Edwards Aquifer-Comal Springs ecosystem provides critical habitat for several endangered and threatened species, including salamanders (the Texas blind salamander and San Marcos Spring salamander), fish (the San Marcos gambusia and fountain darter), and Texas wild rice (Glennon, 2002; Sharp and Banner, 2000). In all, 91 species and subspecies of other organisms that are endemic in this aquifer and its associated springs (Bowles and Arsuffi, 1993; Culver et al., 2000, 2003; Longley, 1986).

Most studies predicting groundwater supply focus on usable water quantities given drought frequencies and recharge. Land use is also important because it influences demand as well as runoff and recharge. As a result of water shortages in San Antonio, regulations controlling development were issued beginning in 1970. These regulations included rules for limiting economic development within the recharge zone. As noted previously, economic development often increases the extent of impervious surfaces that, in turn, cause more rapid runoff and loss of infiltration during and after precipitation events. Studies indicate that when impervious cover exceeds 15 percent of the surface of a watershed, there are adverse impacts on surface water quality and subsurface water recharge (e.g., Veni, 1999).

The quality of groundwater is also an issue. Increasing concerns about water pollution of the Edwards Aquifer led former (now deceased) Congressman Henry B. Gonzalez of San Antonio to propose the Gonzalez Amendment to the Safe Drinking Water Act of 1974. The